FISHERY MANAGEMENT INVESTIGATIONS

IDAHO DEPARTMENT OF FISH AND GAME
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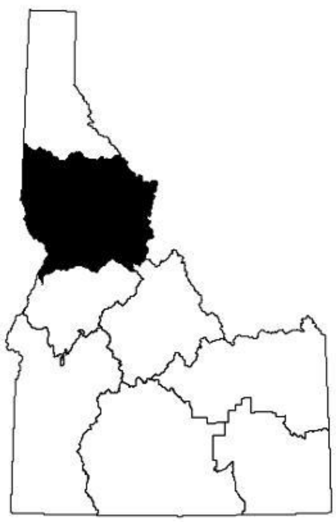

CLEARWATER REGION
2017

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DEER CREEK RESRVOIR: TIGER TROUT EVALUATION

ABSTRACT

Surveys were conducted in Deer Creek Reservoir to evaluate changes in Golden Shiner *Notemigonus crysoleucas* (GS) abundance and length due to stocking catchable size tiger trout (Brown Trout *Salmo trutta* X Brook Trout *Salvelinus fontinalis* TT), and evaluate whether abundance and length of stocked hatchery trout were adequate to meet fishery needs. Surveys were conducted in 2017 using boat-mounted electrofishing of historic transects and compared to findings from previous years. Our results indicate that GS Catch-per-unit-effort (CPUE) was highly variable among years, while mean length increased by 12 mm. For tiger trout, CPUE and mean relative weights declined from 2016 to 2017. Angler total use of TT was low and did not increase from 2016 to 2017 as expected, while angler tag returns are occurring more quickly each year. Additionally, anglers appear to primarily catch TT > 255 mm, indicating ~30% of TT stocked are unavailable to the fishery during the first year. Preliminary analysis suggests that changes in the GS population are likely due its maturing over time, while TT may be experiencing increased intraspecific competition after two years of stocking catchable size fish and high mortality or low carryover from reservoir conditions. However, with only two years of data, additional sampling for GS and hatchery trout (including the collection of otoliths from TT) is needed to provide a more thorough evaluation and determine if changes to our stocking rates and sizes, and regulations should be considered.

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INTRODUCTION

Deer Creek Reservoir (DCR) was constructed during 2003 by impounding Deer Creek, a tributary of Reeds Creek that flows into Dworshak Reservoir. Deer Creek Reservoir is an important part of the region's lowland lake program as it allows trout harvest in an area where all stream fishing is under restrictive harvest regulations (two trout per day). It also adds diversity to our fisheries program as it is the only lowland lake managed solely as a cold-water fishery. It was originally managed with a put-and-take Rainbow Trout *Oncorhynchus mykiss* (RBT) fishery, and put-grow-take Westslope Cutthroat Trout *O. clarkii lewisi* (WCT). Brook Trout *Salvelinus fontinalis (BKT)* were stocked beginning in 2012 as an additional put-grow-take fishery. Deer Creek Reservoir was managed with general trout regulations until 2019, when it was changed to six trout, only two of which may be tiger trout (none less than 356 mm).

Golden Shiner *Notemigonus crysoleucas* (GS) appeared in DCR soon after the reservoir filled. Upon their discovery in 2006 and again in 2010, the reservoir was treated with rotenone in attempts to eradicate this invasive species (Hand 2010; Hand et. al. 2013). The attempts to eradicate GS from DCR were based on several issues: 1) GS are effective planktivores, and an overabundance of GS in DCR could potentially reduce the quality and quantity of the zooplankton available for trout; and 2) we were concerned that GS might spread downstream into Dworshak Reservoir which supports an important kokanee fishery that has been found to generate more than $4 million in fishing-related expenditures annually (IDFG, *unpublished data*).

Unfortunately, GS were again discovered in DCR in 2012. The failure of the rotenone treatments was likely due to a combination of factors including the high level of habitat complexity within the reservoir (large slash piles had been left to provide habitat), springs/seeps that could provide clean water refuge, and GS resistance to rotenone. Golden Shiner have a natural resistance to rotenone and are capable of developing a higher resistance to rotenone which would increase each time the same population is exposed (Orciari 1979). If the initial renovation was not 100% effective, any surviving GS would have the potential of creating a rotenone-resistant population. Additionally, GS were found in several ponds in the nearby Schmidt Creek drainage (near Weippe, Idaho) during the construction of Deyo Reservoir. Nez Perce Tribe fisheries biologists also reported finding GS in nearby drainages including Orofino Creek and Jim Ford Creek. This indicated that GS were widespread, making complete eradication nearly impossible, and making it highly likely that GS had already reached Dworshak Reservoir.

With the realization that rotenone treatments were not effective at eliminating GS, we researched different trout species that could prey upon Golden Shiner, provide desirable fishing opportunities, and not pose a risk to downstream fisheries. Ultimately, we decided to introduce tiger trout (TT; Brown Trout *Salmo trutta* X Brook Trout) into DCR as they had been reported to be an effective predator on minnow species (Sheerer et. al. 1987; Winters 2014), provide desirable fisheries (Winters 2014), and are sterile, posing no risk to downstream fisheries. Our hope was that TT would utilize GS as a prey source, thus improving the food base for trout that depend on zooplankton, and provide a unique fishing opportunity in the region. With this in mind, we began stocking fingerling TT (50 - 75 mm) in DCR in the spring of 2014.

Surveys conducted in 2014 confirmed our concern regarding zooplankton density and quality (i.e. size), as sampling revealed a decline in zooplankton size and density compared to previous years before GS were present (Hand et al. 2017). This decline in food resources may have been a primary reason why only one TT was sampled in 2014 and 2015, and would likely result in future poor growth and survival of trout dependent on zooplankton. Golden Shiner were only present in the stomach contents of Rainbow Trout and Brook Trout > 250 mm. The apparent lack of success in establishing a TT fishery through the fingerling stockings suggested that changes to our stocking strategy were necessary. Decreasing or eliminating the stocking of fingerling trout, and stocking larger trout (TT, RBT, or BKT > 250 mm) could increase their likelihood of survival, decrease the predation pressure on zooplankton, and increase predation of GS. Thus, we changed our strategy to annually stock approximately 2,500 “catchable size” TT (170 - 360 mm) annually. The first catchable TT were stocked in 2016, and sampling occurred in 2016 and 2017 to evaluate the success of this new strategy.

OBJECTIVES

1. Evaluate whether abundance and lengths of Golden Shiner in Deer Creek Reservoir have changed after stocking catchable size tiger trout.

2. Assess whether abundance and sizes of tiger trout, Rainbow Trout, and Brook Trout being stocked into Deer Creek Reservoir are adequate to meet fishery needs.

STUDY AREA

Deer Creek Reservoir is located in Clearwater County, Idaho, 21 km north of Pierce, Idaho (Figure 1). It is a 47.0-ha reservoir located at an elevation of 1,006 m. It has a maximum depth of 11 m, and a maximum volume of 936,000 m^3. All land in the DCR watershed is owned by PotlatchDeltic and is used primarily for timber harvest. Idaho Department of Fish and Game leases the reservoir property. Today, the reservoir is managed to provide family fishing opportunities for cold-water species.

METHODS

Field sampling

Golden Shiner

The GS population in DCR was sampled through boat electrofishing surveys conducted on July 21 and August 22, 2017. Electrical waveform consisted of pulsed DC produced by a Honda EU7000iAT1 generator and a Midwest Lakes Electrofishing Systems (MLES) Infinity pulsator. In order to maintain consistency, we conducted these surveys at the same time of year, and using the same methods as in 2014. Additionally, we sampled the same ten, 50-m transects that have been electrofished in past GS sampling efforts (Figure 2). These transects were originally selected from random shoreline GPS points developed during a vegetation survey conducted in 2012 (Hand et. al. 2016). All transects were sampled during each survey with the boat moving along the shoreline in a clockwise direction. The surveys were conducted at night, and we attempted to net all fish observed. Golden Shiner were measured for total length (mm); weights were not recorded.

Trout

The trout stocked in DCR were sampled through an electrofishing survey conducted on November 1, 2017. Equipment used is described above. In order to maintain sampling consistency, we utilized the same transects that have been used for all trout sampling conducted in DCR (Figure 3). This sampling consisted of one hour of electrofishing, divided into six, 10-minute transects, with fish collected in each transect processed and recorded separately. This allowed a variance to be calculated around the mean catch-per-unit-effort (CPUE), allowing statistical comparisons to the survey conducted in 2016. Electrofishing was conducted along the shoreline in a clockwise direction. The survey was conducted at night, and we attempted to net all trout observed. Trout were identified by species, with total length (mm) and weight (g) recorded for each fish.

A subsample (n = 200) of hatchery catchable-size TT stocked into DCR on June 7, 2017, were tagged at DCR prior to stocking. These fish were randomly selected by netting fish directly from the hatchery truck into a holding tank. All fish selected were tagged with Hallprint model FD-94 anchor tags. Each fish tagged was measured for total length (mm) and weight (g). Tagging data (date, location, species, length, weight, tag number) was submitted to IDFG Nampa Research Office and uploaded to the IDFG "Tag You're It" database.

Data analysis

Golden Shiner

To evaluate whether the abundance of GS changed from 2014 to 2017, we compared the mean CPUE (fish/h) between years for both the June and August surveys using two-tailed t-tests (assuming equal variance) with a significance level of α = 0.10. To evaluate whether the size distribution of GS changed from 2014 to 2017, we compared the mean length of GS sampled between years for both the June and August surveys using two-tailed t-tests (assuming equal variance) with a significance level of α = 0.10.

Trout

To evaluate whether the abundance and sizes of trout stocked into DCR were adequate to meet fishery needs, we assessed how trout density (CPUE), size (mean length), body condition (relative weight), and exploitation changed in relation to changes in stocking practices. Mean CPUE (fish/h) from the electrofishing survey was calculated for TT, RBT, BKT, and WCT and compared with data collected in 2016. This data was not compared to data collected in 2014 or 2015 because these surveys were conducted at a different time of year (i.e. fall vs. spring) or with different gear (gill nets). We compared mean CPUE between 2016 and 2017using a two-tailed t-tests (assuming equal variance) with a significance level of α = 0.10. Mean lengths were compared between years (2016 and 2017) for each species using two-tailed t-tests (assuming equal variance) with a significance level of α = 0.10.

Relative weight (W_r; Wege and Anderson 1978; Neumann et al. 2012) was calculated for each TT tagged in June, and for each TT sampled in November by electrofishing to assess potential changes in body condition from 2016 to 2017 and post-stocking. The relative weight equation is:

$$W_r = \frac{W}{W_s} * 100$$

where W is the observed weight of the fish and W_s is the length-specific standard weight predicted by a weight-length regression. This equation is:

$$log_{10}\ Ws = a + (b * log_{10}\ total\ length)$$

where *a* is the intercept and *b* is the slope of standard weight equations developed for many fish species (Wege and Anderson 1978; Neumann et al. 2012). We used the length-weight relationship for Brown Trout (a = -4.867 and b = 2.96) from Blackwell et al. (2000) and used by Messner and Schoby (2019) for tiger trout evaluations in Wallace Lake, Idaho. The mean relative weight for TT sampled by electrofishing in November of 2017 were compared to TT sampled in November of 2016 using a two-tailed *t*-test (assuming equal variance) with a significance level of $\alpha = 0.10$. The mean relative weight for TT tagged in June, 2017 were compared to TT sampled in November, 2017 using a two-tailed *t*-test (assuming equal variance) with a significance level of $\alpha = 0.10$.

We evaluated angler exploitation of hatchery catchable-size TT stocked into DCR. Calculation of angler exploitation rates and associated CIs (based on reported tags) followed the methodology described in Meyer et al. (2010). Angler exploitation (percent of stocked fish harvested) and total use (percent of stocked fish caught) of TT stocked into DCR on June 7, 2017 were compared to TT stocked in 2016. To facilitate comparisons, 90% CI were calculated around exploitation and total use rates 365 and 730 days post stocking. To show how rapidly the tagged/stocked TT in 2016 and 2017 were caught, this data was displayed graphically with percent of reported tags on the y-axis and days-at-large on the x-axis. To evaluate which size classes of stocked TT were being caught by anglers, we compared mean length of the TT tagged in 2016 and 2017 to the mean length (at time of tagging) of the TT reported as caught by anglers. Ninety-percent CIs were calculated around these mean lengths to assist with comparisons between TT tagged and reported caught.

RESULTS

Golden Shiner

The July CPUE for GS in 2017 (51 fish/transect; ±5) was lower and significantly different than was observed in 2014 whereas the August CPUE in 2017 (95 fish/transect; ±4) was higher and significantly different than 2014 (Figure 4). Golden Shiner mean total length in July (89 mm) and August (77 mm) of 2017 was longer and significantly different than was observed in these months in 2014 (Table 1; Figure 5).

Trout

The 2017 trout survey resulted in the capture of 74 TT, 144 RBT, 6 BKT, and 0 WCT. The mean CPUE for TT was 74 fish/h (±25), lower and significantly different than observed in 2016 (Figure 6). Tiger trout mean total length observed in 2017 was 293 mm (±8), which was not significantly different than observed in 2016 (Table 2; Figure 7).

The mean CPUE for RBT in 2017 was 144 fish/h (±54), higher than observed in 2016 but not significantly different (Table 2; Figure 6). Rainbow Trout mean total length observed in 2017 was 299 mm (±3), longer than observed in 2016 and significantly different (Table 2; Figure 8).

The mean CPUE for BKT in 2017 was 6 fish/h (±3), lower and significantly different than observed in 2016 (Table 2). Brook Trout mean total length was 243 mm (±49), shorter than 2016 but not significantly different (Table 2; Figure 9). No WCT were sampled in the 2017 survey.

Relative weights of TT sampled in November 2017 ranged from 62 to 102, with a mean of 75 (Figure 10). This mean relative weight was lower, and significantly different than for TT sampled in November of 2016 (W_r = 79; α = 0.10; P = 0.001). It was also lower and significantly different than when TT were tagged and stocked in June 2017 (W_r = 98; α = 0.10; P < 0.001; Figure 10).

Angler exploitation and total use of stocked TT through 365 and 730 days-at-large were higher in 2017 than 2016 although the 90% CIs overlapped in all cases (Table 3). Fourteen of the 200 TT tagged in 2017 were reported as being caught within 730 days of being stocked. Twelve (86%) of these fish were caught within 365 days and 11 (79%) were caught within 100 days (Figure 11). Seven of the 99 TT tagged in 2016 were caught within 730 days of being stocked. Four (57%) were caught within 365 days and two (29%) were caught within 100 days (Figure 12).

The mean total length of TT tagged on June 7, 2017 was 271 mm (±3), whereas the mean total length of these TT that were reported as caught was 296 mm (±12) at the time of tagging (Figure 12). About 30% of the TT tagged in 2017 were < 250 mm whereas none of the TT reported as caught were < 250 mm at the time of tagging. The mean total length of TT tagged in 2016 was 286 mm (±5) and the mean total length of these TT that were reported as caught was 287 mm (±24) at the time of tagging (Figure 12). About 14% of the TT tagged in 2016 were < 250 mm and 14% of the TT reported as caught were < 250 mm at the time of tagging.

DISCUSSION

Golden Shiner

Golden Shiner CPUE was highly variable between months within a year and showed conflicting results when trying to evaluate whether GS abundance changed from 2014 to 2017. High variability in CPUE can be due to environmental factors (temperature, weather, water clarity, time of day) that can influence fish distribution and habitat usage, population variation (reproduction, predation), and sampling bias (Byrne et al. 1983; Hubert and Fabrizio 2007; Stockhausen and Fogarty 2007). Diel migrations of GS between littoral and limnetic zones, and vertically in the water column, have been documented (Hall et al. 1979). These migration patterns have been observed in DCR; therefore, sampling at different times of day or weather conditions could have a substantial impact on CPUE. Golden Shiner are highly fecund, and capable of spawning multiple times per year (Lazur and Chapman 1996). Variations in spawn timing and success could have impacts on CPUE as well. In DCR, we have addressed potential sampling bias by standardizing when (same time of day and month) and where sampling occurs, effort, and the type of gear used. As such, sampling variations in DCR are likely due to variability in weather, water clarity, and/or changes in the GS population. When high variability occurs in sampling, it can often be addressed through additional sampling or pooling of available data (Guy and Brown 2007; Stockhausen and Fogarty 2007; Kevin Meyer *personal communication*). We recommend pooling the July and August samples to increase sample size and reduce variability. Pooling data for 2014 and 2017 results in almost identical CPUEs of 72.1 (±21.0) and 72.8 (±11.2). Additionally, with only two years of data, we recommend continuing GS surveys to allow for a better evaluation of trends in CPUE.

The mean length of GS was significantly longer for both months of sampling in 2017 compared to 2014. Predators are known to directly impact prey size structure, often resulting in an increase in average prey length when smaller prey is targeted (Tonn et al. 1992; Persson et al. 1996; Nilsson and Bronmark 2000). This has been shown to occur with GS after the introduction of predators such as Lake Trout *Salvelinus namaycush* and Largemouth Bass *Micropterus salmoides* (Mittlebach et al. 1995; Johannes et al. 2011). Tiger trout have been found to become picivorous at sizes > 270 mm (Miller 2010; Hand et al. 2020). The TT stocked in 2016 and 2017 averaged around 270 mm suggesting a large portion of these fish were of the size where they could immediately prey upon GS. However, the shift in the GS population length frequency toward larger fish may not be exclusively a result of predation. In 2014 the GS population in DCR was relatively new and therefore skewed towards younger fish as the population became established through natural reproduction. In 2017, we would expect mean length to be larger due to multiple years of growth and reproduction. This would explain the increase in mean length and higher abundance of larger fish, but does not account for the decline in abundance of smaller fish. Golden shiner primarily feed upon Daphnia are not known to impact their recruitment through predation on their own young (Hall et al. 1979; Lazur and Chapman) Therefore, it appears that TT are targeting and preying on GS < 80 mm in length; however, we did not sample stomach contents or determine preferred prey sizes during 2017. In eastern Washington lakes, studies indicates that TT prey averaged 64 mm (Miller 2010). Stomach samples of TT were collected in previous surveys on DCR, and while they were in poor condition it appeared TT were mostly consuming GS < 100 mm (Hand et al. 2020). In contrast, mean total length was 139 mm for Utah Chub *Gila atraria* consumed by TT in Scoville Reservoir, Utah, (Winters 2014). While this indicates TT are capable of consuming larger prey than they are in DCR, the TT in Scoville reservoir were larger (mostly > 350 mm). Our data suggests the observed decline in abundance of GS < 80 mm and increased average size is likely due to a combination of a more mature population and predation by TT. It also suggests TT of the sizes present in DCR during 2017 may not be as effective at consuming large GS, as they are likely faster and evasive, or that there are not enough larger TT to impact the abundance of larger GS. We recommend additional sampling for GS, as understanding how TT may be influencing GS size and abundance will be important for developing TT stocking rates that allow fish to grow to sizes that are appealing to anglers yet abundant enough to provide acceptable catch rates.

Trout

Tiger trout mean CPUE was lower and significantly different in 2017, while mean length did not change significantly from 2016 to 2017. The decline in mean CPUE was likely due to a combination of competition, sampling bias, or other unknown factors. Exploitation was not likely a factor, as it is still low despite the increase in 2017. Abundance of TT was higher in 2016 and 2017 compared to surveys conducted in 2014 and 2015 when only fingerling TT were stocked (Hand et al. 2020). In those years, only one TT was collected during any survey. This supports the decision to switch from stocking fingerlings to catchable size TT. The decline in mean CPUE and lack of an increase in mean length from 2016 to 2017 is concerning. If TT are successfully overwintering in DCR, we would expect to see more and larger fish. Understanding the age of the TT in the fishery will allow us to evaluate growth, and mortality and assess whether TT are overwintering and contributing to the fishery for more than one year. We recommend to continue monitoring this TT population, but include collection of aging structure to better assess growth and mortality, and the addition of gill nets to evaluate potential sampling bias.

The mean CPUE for RBT sampled in 2016 and 2017 did not differ significantly, while mean length was shorter and significantly different in 2017. The mean size of stocked RBT can differ by as much as 50 mm due to annual differences in growth, date of stocking, and hatchery of origin (Chris Jeszke, *personal communication*; IFWIS 2021). However, average size at stocking was only 5 mm longer in 2016, indicating this was not a factor in the difference in mean length (IFWIS 2021). The RBT fishery in DCR is put-and-take, and based on previous surveys carryover averages ~6% from year-to-year (Hand et al. 2016). This suggests that few of the RBT sampled in DCR were from previous year's stocking. Thus, the difference in mean length was likely related to variation in growth between years. Further evaluation of angler exploitation of RBT and their age structure would provide information to assess how anglers and annual mortality were influencing the quality of this fishery and whether stocking rates need to be adjusted.

A significant decline in BKT CPUE was observed from 2016 to 2017. Survey data in 2016 suggested stocking fingerling BKT was somewhat successful. The significant decline in 2017 is likely the result of a combination of predation by TT and reduction in stocking rates. In 2013, 10,000 BKT were stocked, which was 3-4 times more than was stocked each year since. Now that DCR has been stocked with catchable sized TT for two years, predation on fingerling BKT has likely increased. Additionally, if angler effort is increasing due to the popularity of TT, increased exploitation of BKT may be occurring as well. Further evaluation of angler exploitation of BKT and their age structure would provide information to assess how anglers and annual mortality were influencing the quality of this fishery and whether stocking rates need to be adjusted. We also recommend the addition of gill nets to evaluate potential sampling bias.

With only two WCT captured in 2016 and none in 2017, few, if any, remain in DCR. Since they were last stocked in DCR in 2011 and have an expected life span of ~6-8 years (Sigler and Zaroban 2018), this was to be expected. We stopped stocking WCT fingerling due to competition for food resources (primarily zooplankton) with GS. We recommend evaluating the zooplankton population in DCR to determine if large zooplankton are abundant enough to support renewed stocking of WCT. However, potential predation by TT would need to be considered as well.

The relative weights of TT declined significantly from when they were stocked in June until they were resampled five months later in November. Similar findings were observed in Wallace Lake, Idaho, and Scofield Reservoir, Utah (Winters 2014; Messner et al. 2017; Messner and Schoby 2019). The decline in TT relative weights in DCR occurred across all size classes, indicating that TT large enough (> 270 mm) to exploit GS did not exhibit improved condition over smaller fish that may rely on other food sources within the first few months in the reservoir. Low relative weights may be indicative of poor food resources or stress (Flickinger and Bulow 1993; Schramm and Willis 2012). We must consider that some decline in relative weights would be expected for fish after leaving a hatchery where food is plentiful. Declines in hatchery trout relative condition after stocking is well documented, and has been attributed to food limitations, stress, and increased movement/activity (Reimers 1963; Ersbak and Haase 1983; Baird et al. 2006). Summer conditions in DCR (low DO and high temperatures) can be stressful for trout, and may have impacted relative weights of TT sampled in the fall. Thus, the observed decline in TT relative weight may be due to the transition from hatchery to wild, the effects of summer reservoir conditions, or intra- or inter-specific competition. However, mean relative weight of TT > 350 (84; more likely to be carryover fish or those more effective at preying on GS) was higher and significantly different than fish < 350 mm (74; $\alpha = 0.10$; $P = 0.018$). This mean relative weight was still lower than for TT at stocking, but indicates that those fish reaching larger sizes are recovering from the initial decline in relative weight post-stocking.

Additionally, mean relative weight of TT sampled in November 2017 was lower, and significantly different from those sampled in November 2016. This could be a result of increased competition or is the result of differences in relative weight of TT stocked in 2016 compared to 2017. Unfortunately, relative weights could not be calculated for TT stocked in 2016, as equipment malfunction prevented fish from being weighed. Average length of TT stocked in 2016 was higher and significantly different than in 2016, so this could have been a factor in the difference in mean relative weight between the two sampling years. This is worth monitoring in future surveys, as a continued decline in mean relative weight would indicate a need to consider reduced stocking rates if our goal is to produce larger fish. With TT relative weights appearing to be low across numerous studies, we should also consider the possibility that these relative weights are actually normal for TT after stocking in reservoirs, and that the use of the length-weight equation for Brown Trout may not be appropriate. If we continue to see similar relative weights in future surveys, we recommend exploring the development of a TT-specific length-weight equation that would be more appropriate for Idaho reservoirs.

The angler total use rate for TT through 365 days in 2017 was 13.3% and did not increase significantly from 2016. These estimates are within the range seen at Wallace Lake, Idaho (10.7 - 20.5%) from 2015 to 2017 (Messner and Schoby 2019), and are similar to RBT stocked in DCR in 2012 (~17%; Hand et al. 2016). However, these total use rates would be considered low, as they are below the Idaho statewide average in 2011 – 2012 (~28%; Cassinelli 2014), and the mean for all Clearwater Region reservoirs in 2012 (~24%; Hand et al. 2016). The lack of change in total use rate from 2016 to 2017 was surprising, as this was the second year of stocking catchable size TT in DCR and word had spread about the quality of this fishery. We expected angler effort of TT and total use to increase in 2017 as this fishery became more popular. As such, angler effort towards TT may not be increasing. The size of TT stocked into DCR could also be an issue. Higher catchability of larger fish has been well documented with stocked trout (Wiley et al. 1993; Cassinelli et al. 2016). However, only one tagged TT < 255 mm (at the time of stocking) from the combined 2016 and 2017 tagging events in DCR was reported caught by anglers, resulting in total use rates of 22% for fish > 255 mm and 4% for TT <255 mm. Therefore, ~30% of the TT stocked in DCR annually are smaller than the size anglers catch and unavailable to the fishery during the first summer. If natural mortality is high during the first year after stocking, angler exploitation may not increase much if we continue to stock TT at a similar length distribution. Stocking TT at larger sizes and/or reducing the proportion of stocked TT < 255 mm would provide more opportunity for anglers. Additional evaluation of exploitation, and sampling TT in the spring prior to stocking, would facilitate our assessment of TT use and carryover.

The timing of when tagged TT were reported as being caught varied considerably between fish stocked in 2016 and 2017. In 2017, reported harvest/catch primarily occurred < 100 days after stocking, whereas tag returns in 2016 were spread out over a two-year period. Tag return studies for stocked catchable size RBT in Idaho indicate that 75% occur within 122 days of stocking (Cassinelli and Meyer 2018), while in Wyoming "most" are caught within two months of stocking (Wiley et al. 1993). This can primarily be attributed to stocking when angling effort is high (spring/early summer) and fishing conditions are good (Wiley et al. 1993; Walters et al. 1997; Cassinelli and Meyer 2018). The tag return data indicated that anglers in DCR caught TT more quickly in 2017 compared to 2016, and that a lower proportion of fish survived to the following year. With few tag returns occurring after early September in 2017, high natural mortality may have occurred around this time, possibly due to low dissolved oxygen from fall turnover. It should be noted that no tags were returned during the gate closure (October 1 - May 20) during either year, suggesting there is limited effort at DCR during this time of year. Preliminary data suggests TT may not be providing as much of a put-and-grow fishery as we had hoped, but primarily a put-and-take fishery. Due to the variability of when fish were caught after the two stocking events and limited understanding on annual mortality, additional exploitation studies coupled with age, growth, and mortality work are needed to assess whether a put-grow-take TT fishery in DCR is a reasonable expectation. We also recommend sampling TT in the spring prior to stocking to better evaluate carryover.

Stocking catchable sized TT in DCR has created a unique fishing opportunity that anglers are beginning to take advantage of. Additional surveys are needed to provide for a more thorough evaluation of this fishery and management directions we should take to improve upon it. Therefore, we recommend conducting additional GS and trout population surveys (including collecting otoliths from TT and GS for age, growth, and mortality data), and angler exploitation surveys of RBT and TT, in order to better understand if changes to our stocking rates and sizes, and regulations should be considered.

MANAGEMENT RECOMMENDATIONS

1. Continue to evaluate Golden Shiner CPUE, and pool data from July and August samples to reduce variability in data.

2. Collect otoliths (or other aging structure) from Golden Shiner in future surveys to better understand their age growth, and mortality and how tiger trout may potentially be influencing this.

3. Collect otoliths from tiger trout and consider sampling in the spring to better evaluate age, growth, mortality, and carryover.

4. Evaluate zooplankton to determine if there have been changes in size and community structure post-introduction of catchable size tiger trout.

5. Evaluate angler exploitation of tiger trout and Rainbow Trout.

6. Stock tiger trout at lengths > 250 mm to provide a desirable fishery for anglers.

7. Evaluate potential mortality related to summer dissolved oxygen and temperature.

8. Try gillnetting to evaluate if this technique provides different results than electrofishing.

Table 1. Comparison of mean length (mm) between 2014 and 2017 during July and August for Golden Shiner collected by electrofishing Deer Creek Reservoir, Idaho. Significance was set at α = 0.10.

Date	Length	90% CI	P-value
July 2014	69	1	<0.001
July 2017	89	1	
Aug 2014	61	1	<0.001
Aug 2017	77	1	

Table 2. Comparison of mean length (mm) between 2016 and 2017 for trout collected by electrofishing Deer Creek Reservoir, Idaho. Significance was set at α = 0.10.

Species	Date	Length	90% CI	P-value
Tiger trout	2016	292	1	0.200
	2017	293	1	
Rainbow Trout	2016	311	2	<0.001
	2017	299	3	
Brook Trout	2016	285	20	0.150
	2017	243	49	
Westslope Cutthroat Trout	2016	266	48	n/a
	2017	---	---	

Table 3. Angler exploitation (harvested fish) and total use (harvested and released fish) of tiger trout stocked into Deer Creek Reservoir, Idaho, in 2016 and 2017, based on angler-reported, T-bar anchor tags through 365 and 730 days-at-large.

			Disposition						
Tagging date	Tags released	Days at large	Harvested	Harvested b/c tagged	Released	Adjusted exploitation	90% CI	Adjusted total use	90% CI
6/22/2016	99	365	4	0	0	9.8%	7.4%	9.8%	7.4%
		730	7	0	0	14.6%	9.1%	17.1%	9.8%
6/7/2017	200	365	12	0	1	12.1%	6.0%	13.3%	6.3%
		730	13	0	1	15.7%	6.9%	16.9%	7.1%

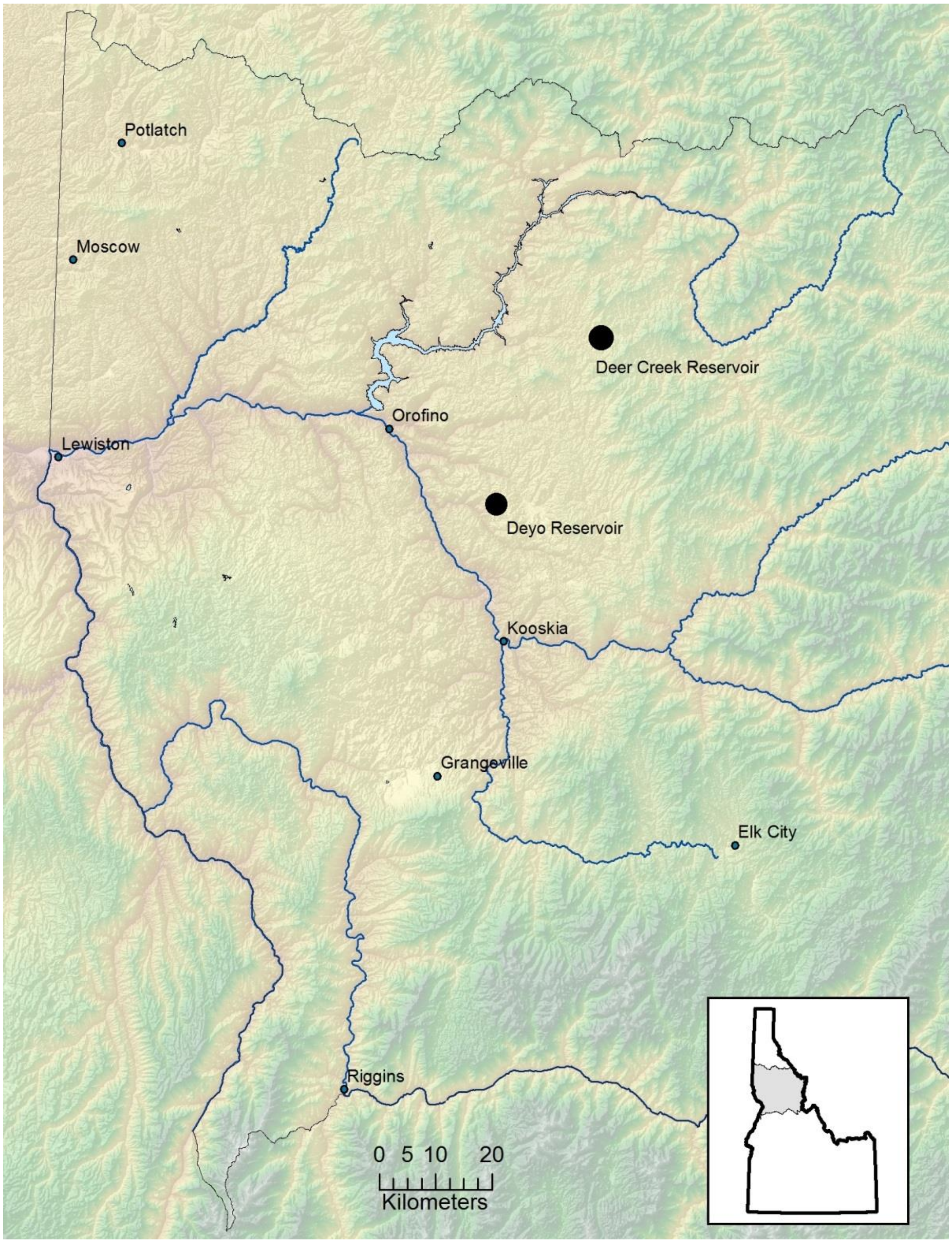

Figure 1. Reservoirs surveyed in the Clearwater Region, Idaho, during 2017.

Figure 2. Locations of starting points for 50-m electrofishing transects used for Golden Shiner surveys in Deer Creek Reservoir, Idaho, in 2014 and 2017.

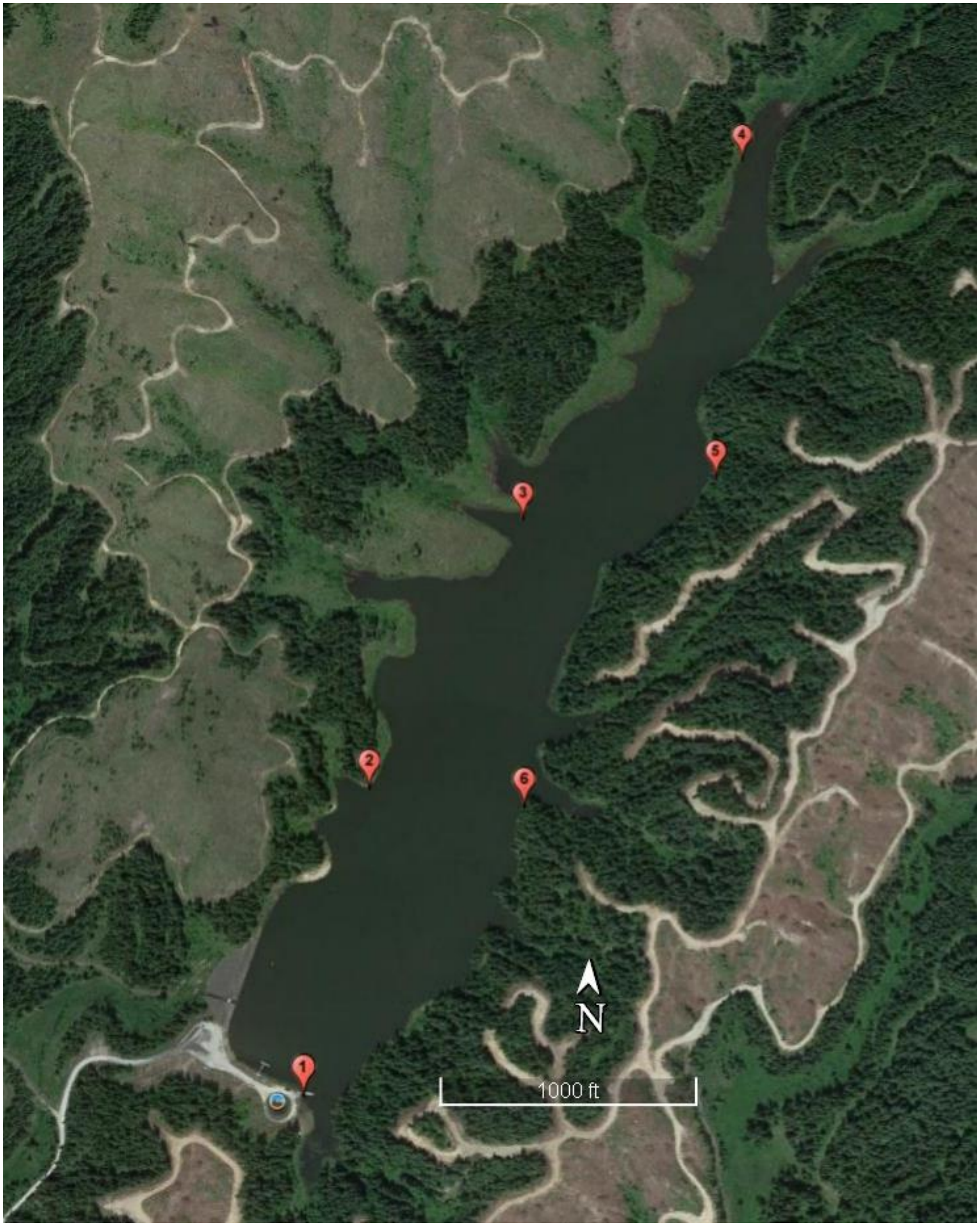

Figure 3. Locations of starting points for trout electrofishing transects on Deer Creek Reservoir, Idaho, in 2016 and 2017.

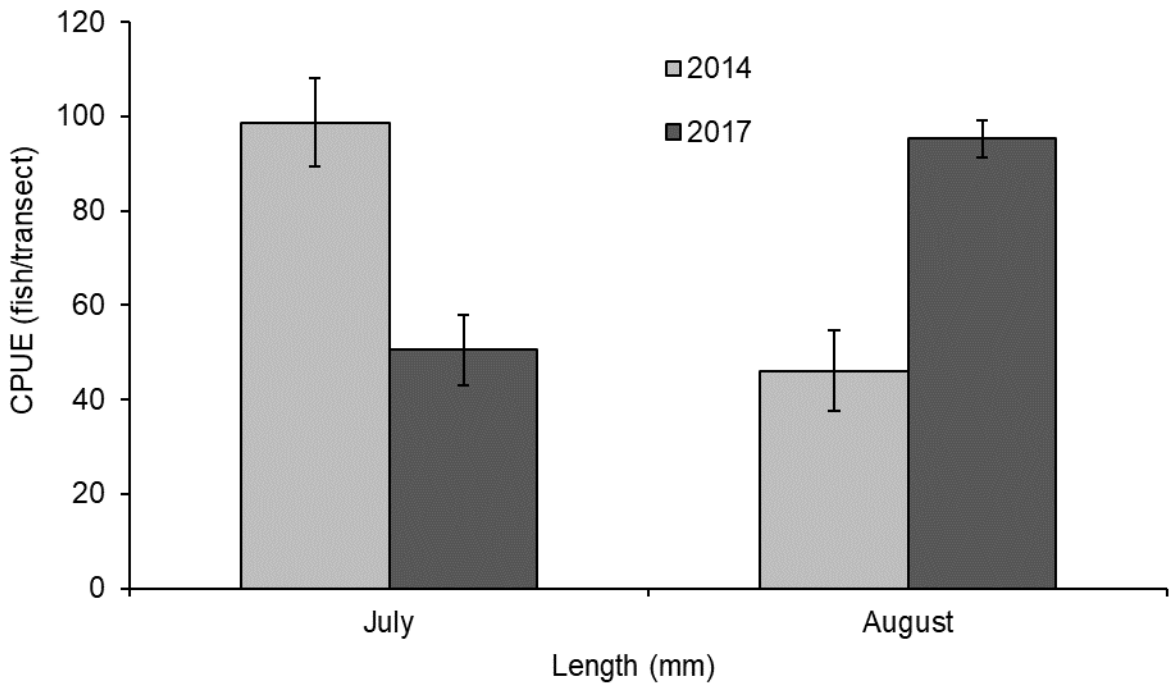

Figure 4. Comparisons of CPUE of Golden Shiner sampled by electrofishing Deer Creek Reservoir, Idaho, in July and August, 2014 and 2017. Error bars represent 90% CIs.

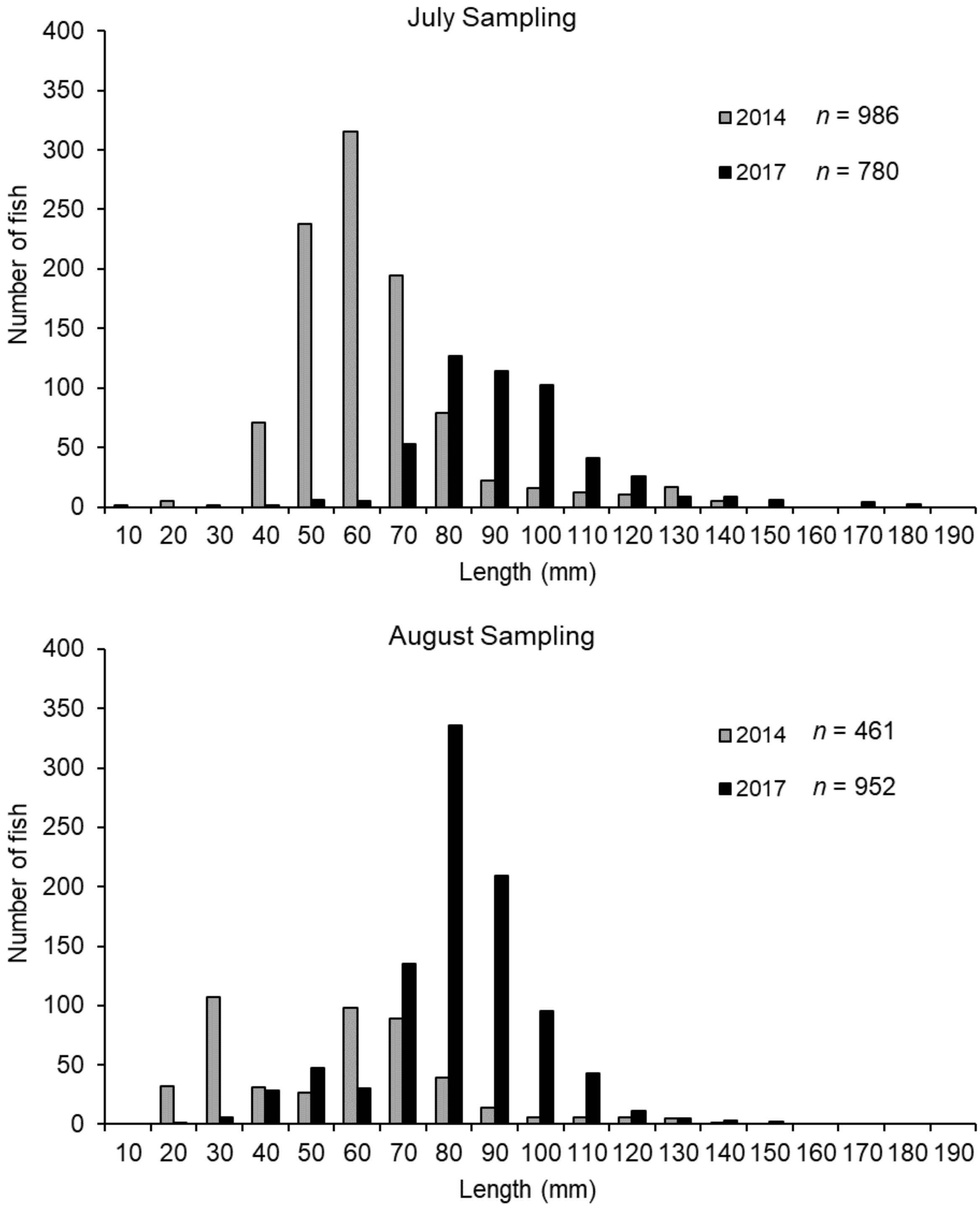

Figure 5. Length-frequency distribution of Golden Shiner sampled by electrofishing Deer Creek Reservoir, Idaho, in July and August, 2014 and 2017.

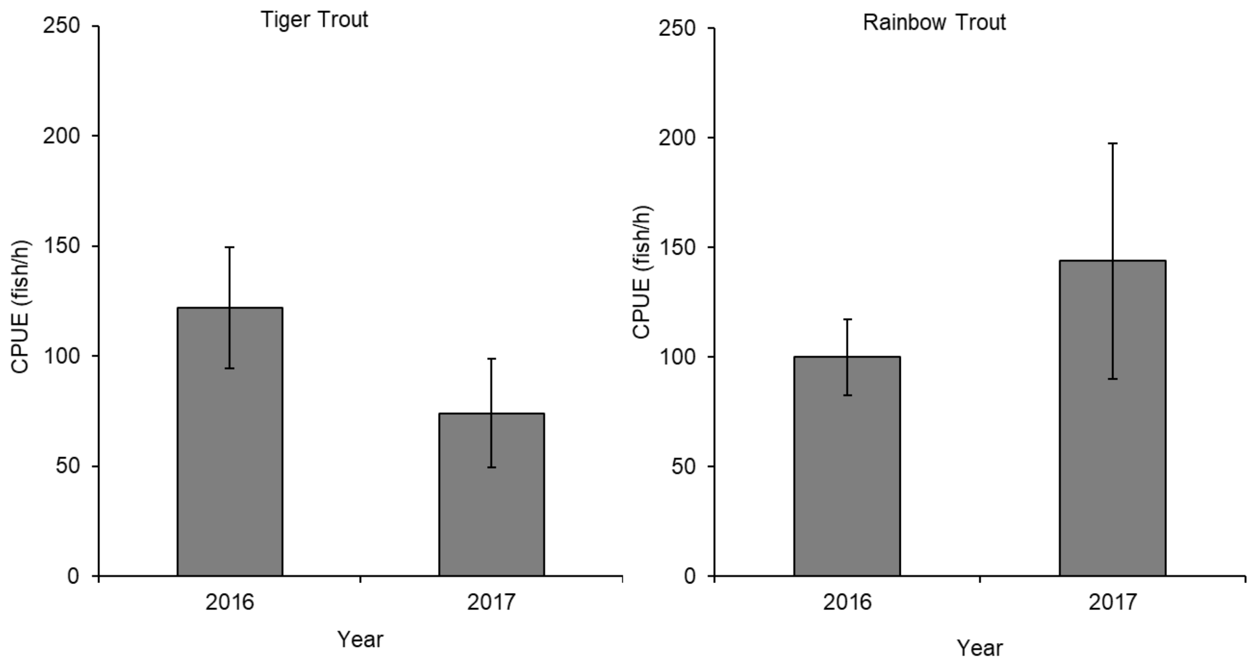

Figure 6. A comparison of CPUE between 2016 and 2017 for tiger trout and Rainbow Trout sampled by electrofishing Deer Creek Reservoir, Idaho. Error bars represent 90% CIs.

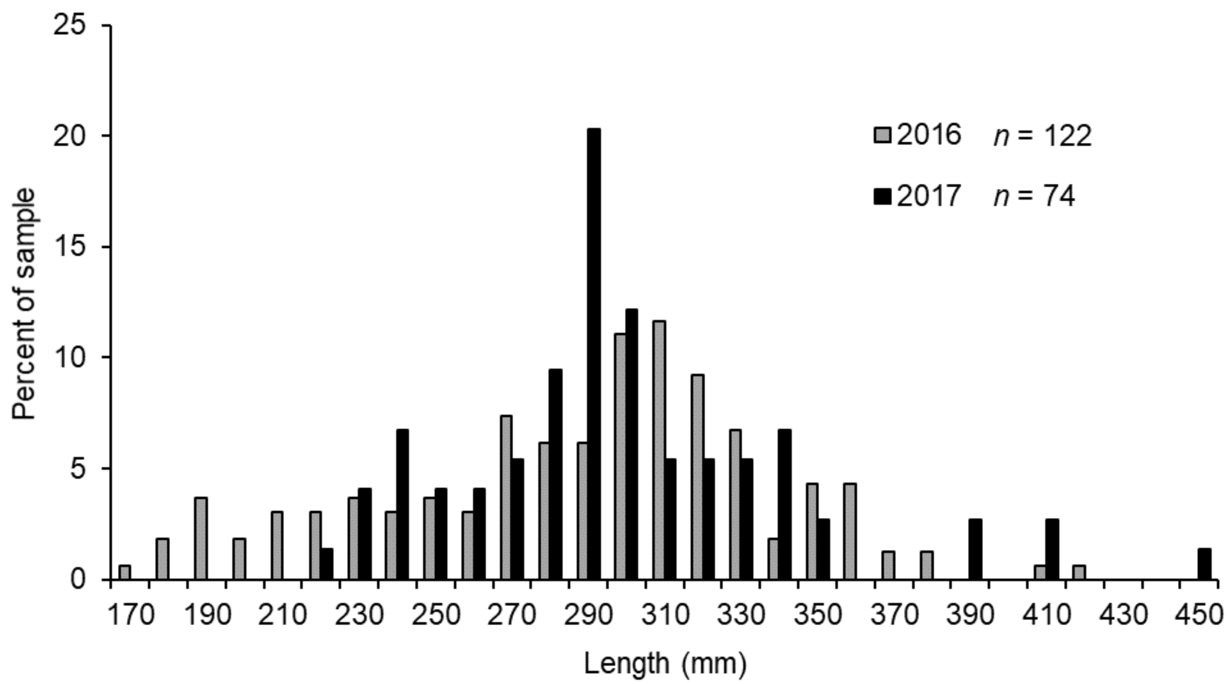

Figure 7. Length-frequency distribution of tiger trout sampled by electrofishing Deer Creek Reservoir, Idaho, in 2016 and 2017.

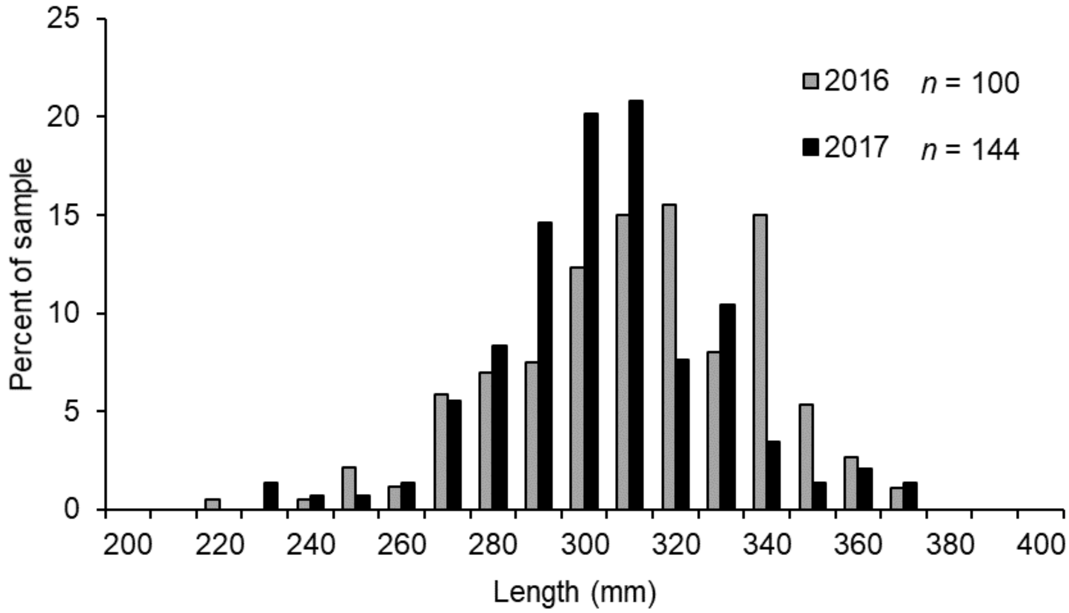

Figure 8. Length-frequency distribution of Rainbow Trout sampled by electrofishing Deer Creek Reservoir, Idaho, in 2016 and 2017.

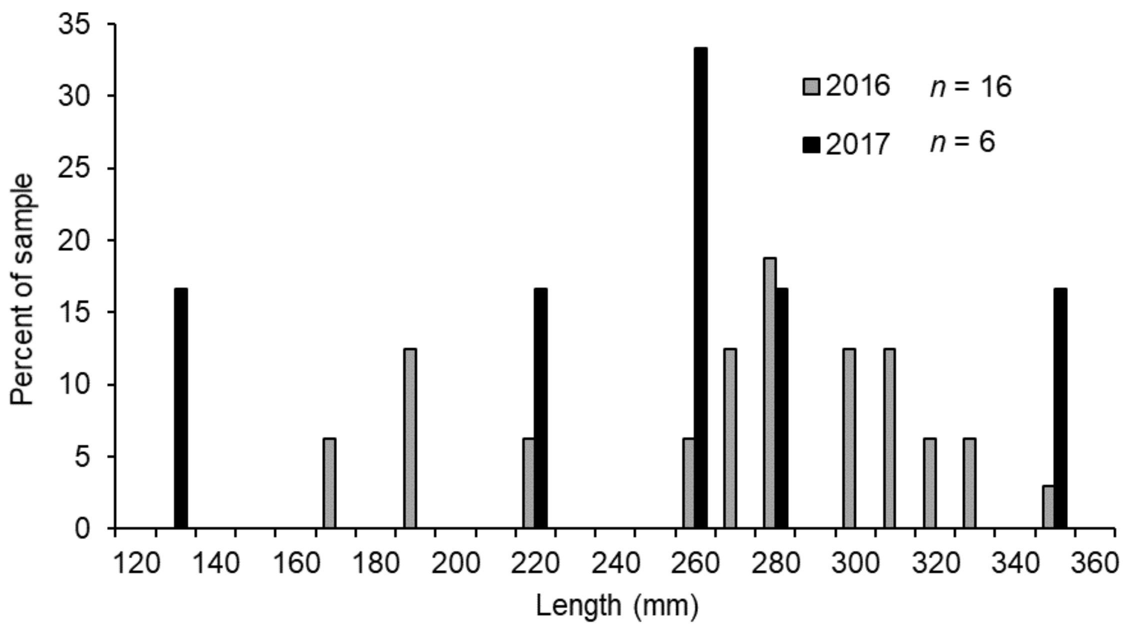

Figure 9. Length-frequency distribution of Brook Trout sampled by electrofishing Deer Creek Reservoir, Idaho, in 2016 and 2017.

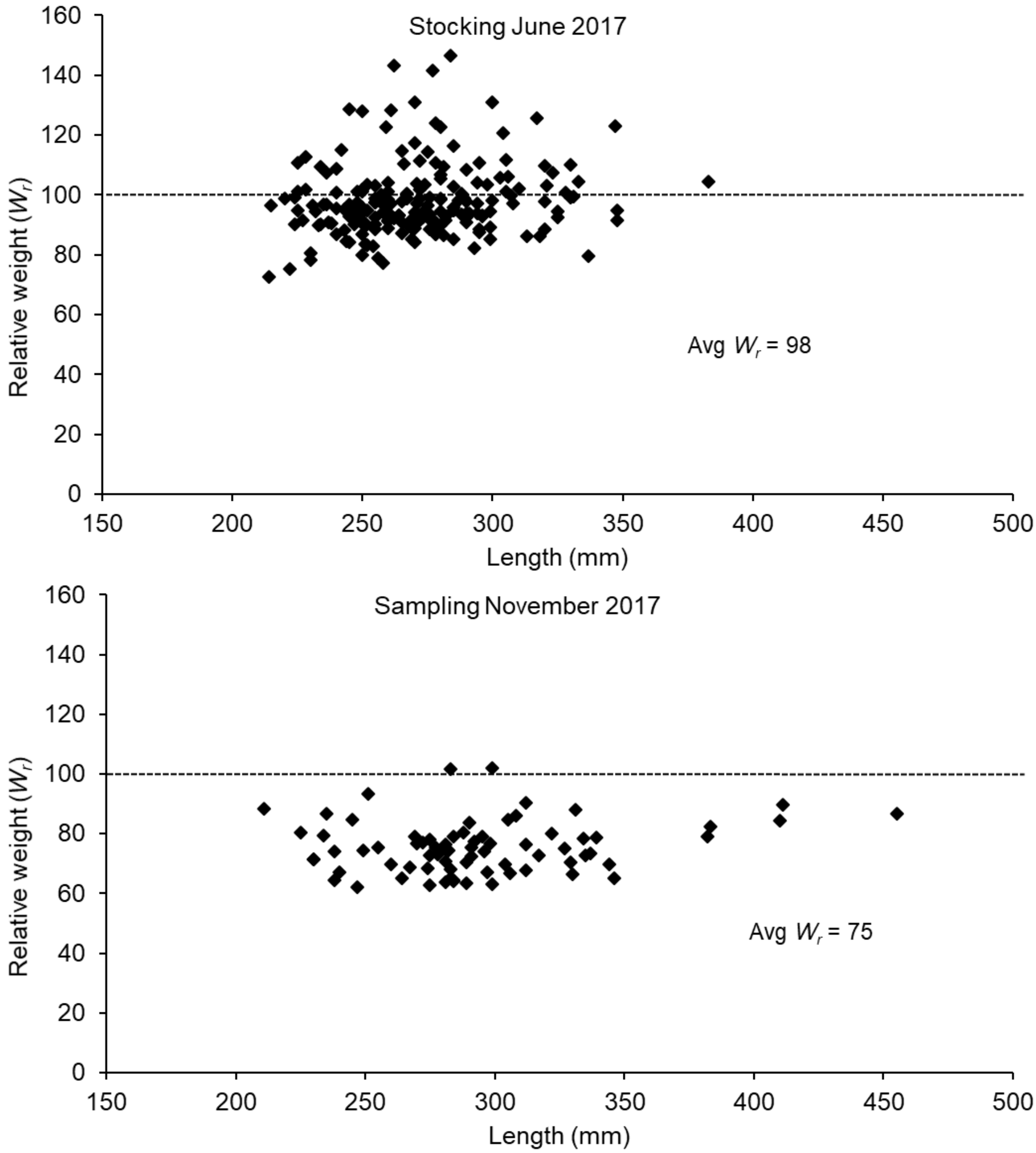

Figure 10. Relative weight of tiger trout tagged and stocked into Deer Creek Reservoir, Idaho, in June, 2017, and those sampled by electrofishing in November, 2017.

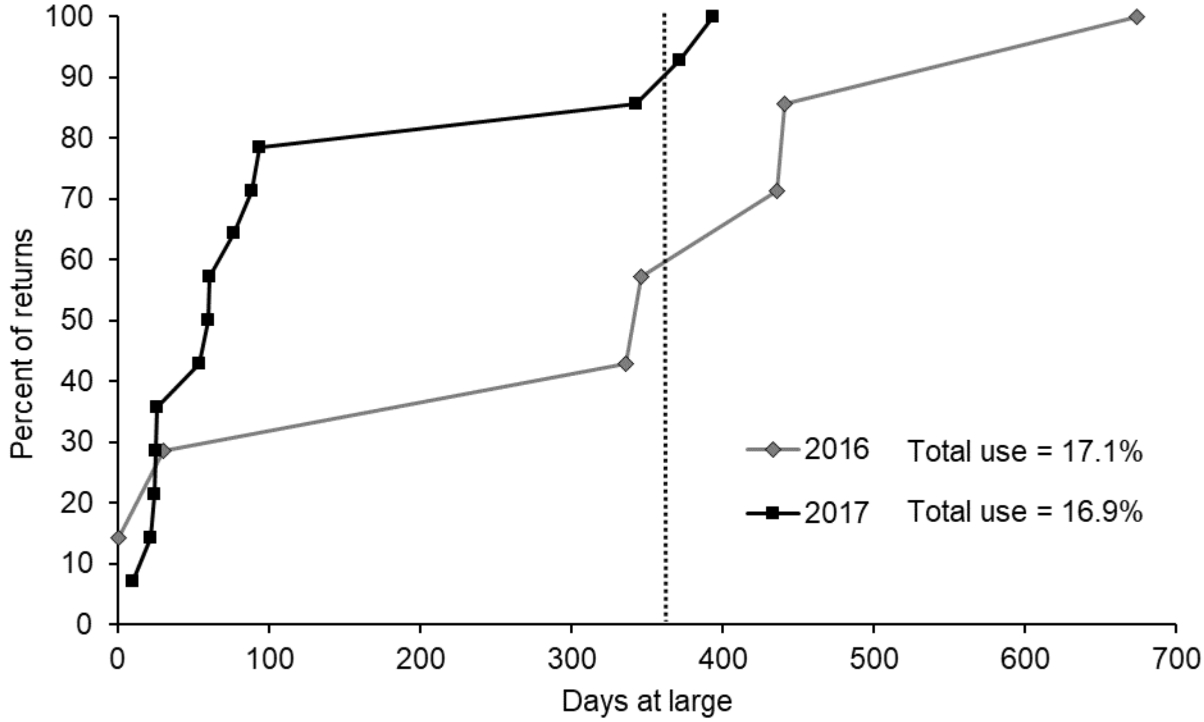

Figure 11. Days-at-large for tagged tiger trout reported as caught by anglers through the Tag You're It program, for fish tagged and stocked in 2016 (n = 7) and 2017 (n = 14) in Deer Creek Reservoir, Idaho, through 730 days-at-large. Vertical dashed line represents 365 days-at-large (1 year). Total use rate is through 730 days-at-large.

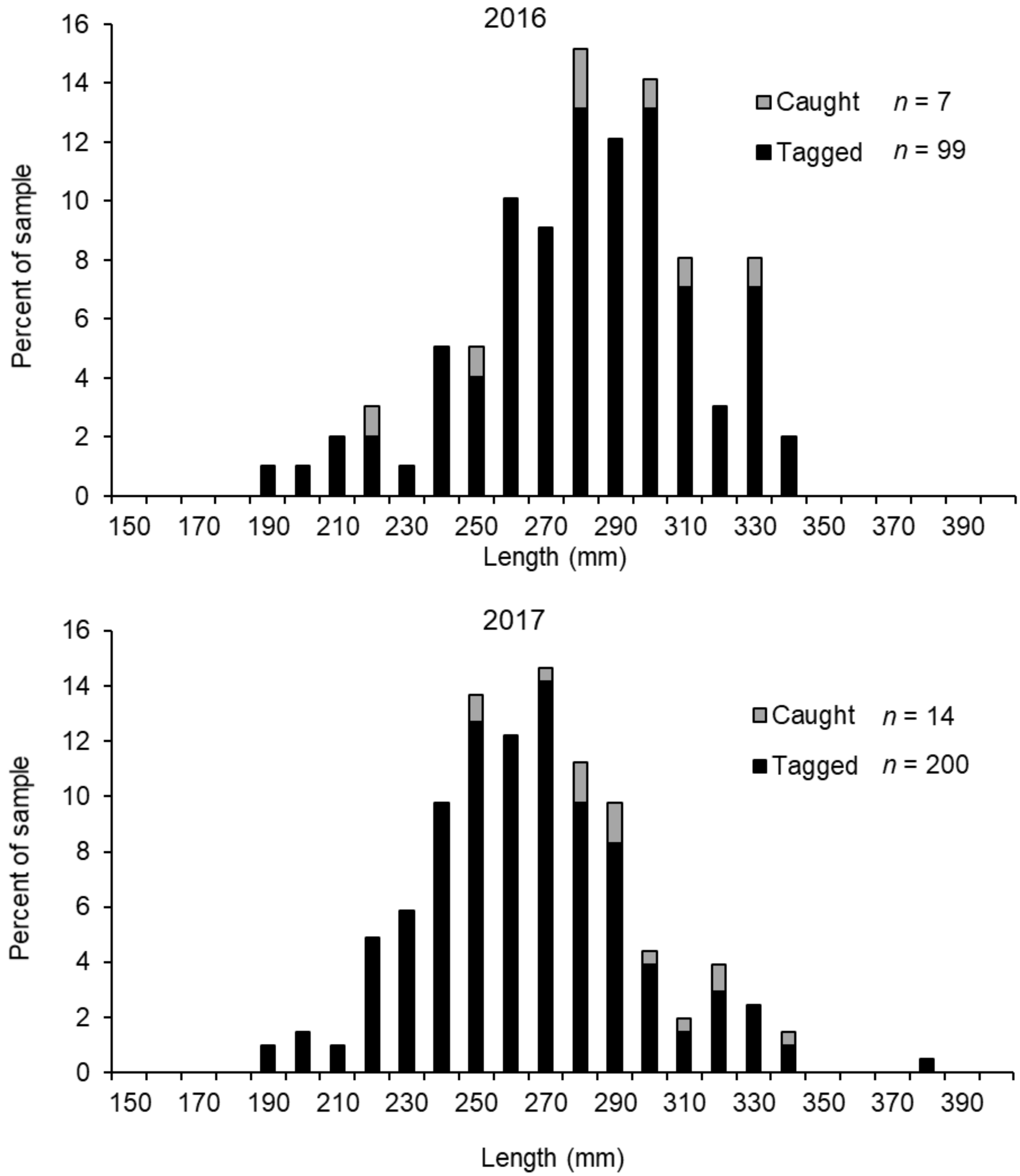

Figure 12. Length-frequency distribution of tiger trout tagged in 2016 and 2017 in Deer Creek Reservoir, Idaho, in 2016, with proportion of each size class caught by anglers (based on angler reported tags).

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DEYO RESERVOIR FISHERY EVALUATION

ABSTRACT

An electrofishing survey was conducted on Deyo Reservoir in 2017 to evaluate whether Largemouth Bass *Micropterus salmoides* (LMB) translocation and restrictive regulations implemented in 2016 improved LMB size structure and recruitment, and the predator:prey balance. The proportion of stock-sized LMB sampled in 2017 increased compared to previous years. The abundance of quality-size LMB increased as well, though to a lesser extent. In contrast, LMB and Bluegill *Lepomis macrochirus* (BG) recruitment appeared to be poor. Relative weights of LMB indicate that food resources are likely limited for fish < 300 mm in length. Relative weights were near 100 for BG of all lengths, indicative of a population in “good” condition. Proportional size distributions were below the thresholds of balance for both LMB and BG. This was due to the recent LMB translocation and because naturally recruited LMB and BG have just entered the stock size range. Preliminary analysis suggests that the restrictive regulations and translocation of LMB have been partially successful, with improvements to the abundance and size structure of the LMB population. However, the translocated LMB did not grow as expected to help fill in the gap in fish > 250 mm. The lack of recent recruitment in both LMB and BG is concerning, and if recruitment of either species does not improve, additional supplementation will be necessary. With only one year of post-treatment data, additional surveys are needed to thoroughly evaluate the LMB translocation and regulation changes.

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INTRODUCTION

Deyo Reservoir was constructed in 2012 to provide a new recreational fishery and an economic boost to the local economy (DuPont 2011). The management strategy for Deyo Reservoir is to provide a "two-story" fishery, with both cold and warm-water species. This includes annual stocking of catchable-size (~ 300 mm) hatchery Rainbow Trout *Oncorhynchus mykiss* (RBT), and managing for balanced and self-sustaining populations of Largemouth Bass *Micropterus salmoides* (LMB) and Bluegill *Lepomis macrochirus* (BG). Largemouth Bass and BG were first introduced in 2012, when 100 LMB (255 - 403 mm) and 350 BG (98 - 177 mm) were translocated from Winchester Lake and Spring Valley Reservoir into Deyo Reservoir. Electrofishing surveys in 2014 and 2015 sampled few LMB > 300 mm, while BG were plentiful but small (averaged ~87 mm). Additionally, reports from anglers and the campground host at Deyo Reservoir indicated that LMB harvest was higher than anticipated. Based on LMB growth rates in nearby reservoirs we anticipated it would take five or six years before the first naturally produced LMB would exceed 300 mm where they could start spawning and become more effective predators on BG (Hand et al. 2016). Therefore, we were concerned that angler harvest could greatly reduce the number of larger size classes of LMB in the reservoir to and significantly delay the amount of time it would take to develop balanced LMB and BG populations.

Restrictive regulations on LMB (daily limit of 2, none > 406 mm) were implemented on Deyo Reservoir in 2016 in an effort to reduce harvest of LMB that were large enough to spawn and prey upon BG, and to improve the predator:prey balance. Minimum length limits are recommended for fish populations that exhibit low rates of recruitment and natural mortality, good growth rates, and high fishing mortality (Novinger 1984; Wilde 1997). They are generally used to protect the reproductive potential of fish populations, prevent overexploitation, increase angler catch rates, and promote predation on prey species (Noble and Jones 1993; Maceina et al. 1998; Paukert et al. 2002; Iserman and Paukert 2010).

Additionally, 313 LMB were translocated from Bonner Lake and Smith Lake near Coeur d'Alene, Idaho, to Deyo Reservoir in June, 2016 to increase the number of LMB in the reservoir (Hand et al. 2020). This was done after the fishery survey conducted in 2016. Supplemental stocking of LMB has been used to control overabundant prey populations, and improve reproductive potential (Hoxmeier and Wahl 2002). We hoped these management strategies would improve LMB size structure and result in more large fish capable of preying upon BG. Surveys in 2017 were designed to evaluate the effectiveness of the new regulations and translocation efforts in creating a balanced warm-water fishery. Specifically our objectives for this study were as follows:

OBJECTIVES

1. Evaluate whether translocation of Largemouth Bass into Deyo Reservoir in 2016 increased the abundance of medium and larger sized (> 250 mm) Largemouth Bass in 2017.

2. Evaluate whether the size structure and recruitment of Largemouth Bass increased after the new Largemouth Bass size restriction and limit (2 fish, none < 406 mm) was implemented in 2016.

3. Evaluate whether the size structure and recruitment of Bluegill has changed in Deyo Reservoir following translocation of bass into it and new Largemouth Bass limits were implemented in 2016.

4. Evaluate the predator:prey balance between Bluegill and Largemouth Bass to determine if there is a need to implement additional management strategies.

STUDY AREA

Deyo Reservoir was constructed in 2012 by damming Schmidt Creek, a tributary to Lolo Creek, Idaho. The reservoir is located approximately 5 km west of Weippe, Idaho, at an elevation of 920 m (Figure 1). At full pool, the surface area of the reservoir is 22.3 ha. It has a maximum depth of 10 m, a mean depth of 5 m, and a maximum volume of about 678,000 m^3. The upper end of the reservoir has been developed into a wetland area to provide habitat for waterfowl and other wildlife. The drainage basin is composed of a mix of forest and cropland. Facilities at the reservoir include a campground with both full hookups and primitive sites, numerous fishing docks (including ADA accessible), boat ramp, picnic pavilion, and toilets.

METHODS

Field sampling

The LMB and BG populations were sampled through an electrofishing survey conducted on June 13, 2017. Boat-mounted electrofishing was performed using pulsed DC from a Honda EU7000iAT1 generator and a Midwest Lakes Electrofishing Systems (MLES) Infinity pulsator. In order to maintain sampling consistency, we utilized the same transects that have been used for all fish population sampling conducted in Deyo Reservoir (Figure 13). This sampling consisted of one hour of electrofishing, divided into six, 10-minute transects, with fish collected in each transect processed and recorded separately. Electrofishing was conducted along the shoreline in a clockwise direction. The survey was conducted at night, and we attempted to net all fish observed. Species, total length (mm), and weight (g) were recorded for each fish sampled. Approximately 10 scales were collected from LMB (n = 35) and BG (n = 34). They were removed from above the lateral line, slightly behind the dorsal fin. Scales were placed in a coin envelope with species, length (mm), weight (g), date, and lake name written on each envelope.

Data analysis

To evaluate the influence of the LMB translocation and restrictive regulations on LMB and BG size structure and recruitment, and predator:prey balance in Deyo Reservoir, we compared CPUE (fish/h), lengths, Proportional Size Distribution (PSD), and relative weights collected in 2017 with surveys conducted prior to the LMB translocation and regulations change (2014 - 2016).

Mean CPUE was calculated for the survey conducted in 2017. Significant differences in mean CPUE among surveys (2014 - 2017) were evaluated using a single-factor ANOVA with a significance level of $\alpha = 0.10$. Post hoc tests to determine significant differences between years were performed using a Tukey-Kramer Test with a significance level of $\alpha = 0.10$.

Length-frequency graphs were developed for LMB and BG sampled in 2017 and compared to previous years (2014 - 2016) to help understand whether there have been shifts in certain size classes over time. In addition, mean lengths were calculated for each year, and we used a single factor ANOVA with a significance level of $\alpha = 0.10$ to test whether mean lengths differed among years. Post hoc tests to determine significant differences between years were performed using the Tukey-Kramer test with a significance level of $\alpha = 0.10$. Mean length for LMB translocated into Deyo Reservoir in 2016 was compared to the mean length surveyed in 2017 using a two-tailed *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.10$.

We evaluated age of LMB and BG from scale samples to assess the effects of the LMB translocation and restrictive regulations on these populations, and to evaluate whether changes in abundance of larger LMB could be attributed to translocation or natural reproduction. The software ImageJ was utilized to mark annuli and measure distances between them. Age was estimated by counting annuli. Age-frequency graphs were developed for each species. Due to a data entry error, we were not able to assign aged scales to the correct fish length. Thus, back-calculated lengths at age, and catch curves for estimating mortality could not be calculated.

The Proportional Size Distribution (PSD; Guy et al. 2007; Neumann et al. 2012) was calculated for LMB and BG and compared to previous surveys (2014 - 2016). Proportional size distribution was calculated for LMB and BG using the following equation:

$$PSD = \frac{\#\ fish \geq qualitysize}{\#\ fish \geq stocksize} * 100$$

stock size and quality size correspond to lengths considered to be the minimum size at which anglers will first catch the species (stock), and consider the fish to be of desirable size (quality). These lengths are 200 mm and 300 mm for LMB, and 80 mm and 150 mm for BG (Gablehouse 1984; Neumann et al. 2012). Proportional Size Distribution values of 40 - 70 for LMB and 20 - 40 for BG are considered to be indicative of balance (Anderson 1980). Proportional Size Distribution for LMB and BG was compared to previous surveys (2014 - 2016) as higher or lower.

Predator:prey model plots were developed by plotting the PSD of LMB (predator, x-axis) against the PSD of BG (prey, y-axis) for surveys conducted from 2014 to 2017. The PSD values for LMB and BG can each fall into three categories: low, desirable, or high. Thus, there are nine possible predator:prey PSD size structure scenarios. Explanations for each situation and recommended management actions are detailed in Schramm and Willis (2012). Annual predator:prey balance was assessed by comparing the location of the PSD plots from year to year.

Relative weight (W_r; Wege and Anderson 1978; Neumann et al. 2012) was calculated for each LMB and BG sampled to assess potential changes in body condition compared to previous surveys (2014 - 2016). The relative weight equation is:

$$W_r = \frac{W}{W_s} * 100$$

where W is the observed weight of the fish and W_s is the length-specific standard weight predicted by a weight-length regression. This equation is:

$$log_{10}\ Ws = a + (b * log_{10}\, total\ length)$$

where *a* is the intercept and *b* is the slope of standard weight equations developed for many fish species (Wege and Anderson 1978; Neumann et al. 2012). To help evaluate whether relative weight changes based on the length of the fish, a scatterplot depicting the relative weight and corresponding length of each fish sampled was developed for each species. To evaluate whether mean relative weight differed among years, we used a single factor ANOVA with a significance level of α = 0.10. Post hoc tests to determine significant differences between years were performed using the Tukey-Kramer test with a significance level of α = 0.10.

RESULTS

Largemouth Bass

The mean CPUE for LMB was 60 fish/h (±10; Table 4). This was the highest mean value since surveys began in 2014 (Figure 14); however, there was no significant difference in CPUE among years (Table 4).

Largemouth Bass mean total length was 264 mm (±18;Table 4). This was larger and significantly different than all other survey years (Table 4; Figure 15). The mean length of LMB surveyed in 2017 was also larger and significantly different (P = <0.001; α = 0.10) than the mean length of LMB translocated into Deyo Reservoir in 2016 (196 mm; Figure 16). The minimum length of LMB surveyed increased from 50 mm in 2014 to 160 mm in 2017 (Figure 16).

Largemouth Bass ranged in age from 2 to 11 years (Figure 17). The majority of LMB aged were 4 - 7 years old, indicating the presence of both natural reproduction and translocated fish.

Proportional Size Distribution of LMB was 34, below the range considered indicative of a balanced population (Table 4). This was lower than 2016, but similar to 2014 and 2015 (Table 4).

Mean relative weight of LMB was 88 (Table 4). This was the lowest for surveys since 2014. However, there were no significant differences among years (Table 4). Relative weights were higher for larger individuals (Figure 18).

Bluegill

The mean CPUE for BG was 724 fish/h (±255; Table 5). This was the lowest since surveys began in 2014 (Figure 14). However, there was no significant difference among years (Table 5).

Bluegill mean total length was 109 (±1; Table 5). This was larger, and significantly different than all other survey years (Table 5; Figure 19). Additionally, mean total length has been higher, and significantly different in each successive year of sampling. The length frequency distribution of BG has shifted towards larger sizes each year (Figure 20). No BG < 70 mm have been surveyed since 2015 (Figure 20).

Bluegill ranged in age from 3 to 5 years old (Figure 21). These fish were all natural reproduction, as BG originally translocated into Deyo Reservoir would be > 5 years old.

Proportional size distribution of BG was 2, below the range considered indicative of a balanced population (Table 5). This was similar to previous surveys.

Mean relative weight of BG was 100 (Table 5). This was similar to 2016 and not significantly different (Table 5). However, it was lower, and significantly different than both 2014 and 2015. Relative weights of BG were evenly distributed around 100 with no differences based on fish length (Figure 22).

Predator:prey balance

Based on the PSD values for LMB and BG, the predator:prey decision model was located in Cell 7 for 2017 (Figure 23). The predator:prey decision model was also located in Cell 7 in 2014 and 2015, while it was located in Cell 9 in 2016.

DISCUSSION

Largemouth Bass

Abundance of LMB was similar to previous years based on CPUE data. This suggests that mortality (natural and angler) and a lack of recruitment likely offset any potential increase in abundance that would have been gained from the translocation and restrictive regulations implemented in 2016 (Maceina and Pereira 2007; Miranda and Bettoli 2007). We are not aware of specific electrofishing catch rates that would indicate when a water body has a sufficient abundance of LMB; however, a CPUE of > 50 fish/h for stock-size LMB is considered a good estimator (Schramm and Willis 2002). For Deyo Reservoir, stock-size LMB CPUE was 51 fish/h in 2017, suggesting that abundance of larger fish may be approaching desired levels.

The overall size structure of LMB in 2017 shifted to longer sizes than observed in previous surveys. The increase in the abundance of stock-sized LMB in the reservoir can be attributed to the translocation efforts and growth of naturally recruited fish. Approximately 99% of the translocated fish were < 300 mm in length, which directly affected the proportion of stock size fish (Hand et al. 2020). This is confirmed by the decline in LMB PSD from 2016 to 2017. Annual growth rates of LMB in other reservoirs (at a similar elevation) in the Clearwater Region typically are around 50 to 70 mm (Hand et al. 2016). This growth rate would account for the 130 - 200 mm size-class of LMB observed in 2016 reaching 190 - 240 mm in 2017. Interestingly, it does not appear that the translocated fish grew much in their first year, as only one LMB between 240 and 310 mm was sampled in 2017. This is the size range that many of the translocated fish should have grown into if they grew 50 - 70 mm over the year. The age data corroborates this as many of the LMB in the 190 - 240 mm size class appeared to be six to seven years old (the naturally recruited fish would be four to five years old).

The increase in in the abundance of quality-sized (> 300 mm) LMB observed in 2017 suggests that the 16 inch minimum length limit implemented in 2016 is making a difference. Not only did we sample more fish > 300 mm than in past surveys, but we also sampled the largest fish we ever observed. Other studies have also shown that minimum length limits can be effective in increasing the abundance and overall size in LMB populations (Paragamian 1982; Novinger 1987; Isermann 2007). With only one post-treatment year of data, additional population surveys are needed to more thoroughly evaluate the LMB translocation and regulation changes. This should include the collection of spines to evaluate age, growth, and mortality of LMB. Additionally, we recommend evaluating angler effort, harvest, and satisfaction to assess their impacts on the success of this project.

Little recruitment of LMB appears to have occurred for the last 3 years (since 2014) based on the ages (few < 4) and sizes (none < 150 mm) of fish captured in 2017. Largemouth Bass recruitment failure has been found to occur in small impoundments due to egg and fry predation and competition for food resources from smaller fish (Anderson and Weithman 1978; Guy and Willis 1991; Aday and Graeb 2012). Additionally, LMB can experience highly variable recruitment, especially in northern waters where poor growing seasons can reduce over-winter survival of young-of-the-year fish (Paragamian 1982; Rieman 1984; Novinger 1987; Parkous and Wahl 2002). Deyo Reservoir appears to contain suitable spawning and rearing habitat, unusually cold years did not occur from 2014 to 2017, and there does not appear to be a lack of spawners, as over 25% of the LMB population in Deyo Reservoir were > 350 mm. Gear bias can also be an issue; however, electrofishing has proven to be an effective sampling technique for smaller LMB and we collected numerous LMB < 100 mm from Deyo Reservoir in 2014. Thus, the poor LMB recruitment in Deyo Reservoir is likely a result of predation on eggs and/or fry from the BG population. After BG were introduced into Deyo Reservoir in 2012, their population quickly exploded. The timing of when this occurred matches up with when LMB recruitment failures began.

Relative weights of LMB indicate that food resources in Deyo Reservoir are limited for fish < 300 mm (Pope and Kruse; Willis et al. 2010). Largemouth Bass are planktivores at sizes < 70 mm, when they primarily switch to piscivory (Huskey and Turingan 2001). Studies have shown that LMB at lengths of 100 - 300 mm prefer to consume Centrarchids 20 - 90 mm in length (Hambright 1991; Hoyle and Keast 1987). The lack of BG in Deyo Reservoir within this size range indicates a lack of food resources for LMB < 300 mm and explains the lower relative weights. While we could not find specific examples in the literature, we could expect there to be potential issues with food resources in newer reservoirs until food webs become established and prey populations mature.

Bluegill

Bluegill CPUE has not changed significantly over the last three surveys. The lack of population growth is likely the result of a combination of recruitment failure and natural mortality over the last few years. However, BG CPUE in Deyo Reservoir was still much higher than the national average CPUE (238 fish/h) reported for small impoundments (Bonar et al. 2009), and the average CPUE (415 fish/h) for Clearwater Region reservoirs in 2012 (Hand et al. 2016). This indicates that densities are high. Since we were unable to properly back-calculate lengths at age, we recommend collecting aging structures again in 2019. Growth rate information will help us assess whether the population is overpopulated.

Mean length of Bluegill has increased during each survey since 2014. This is expected for a recently introduced population (translocated in 2012), and mean length should continue to increase as the population matures (Maciena and Pereira 2007). Bluegill PSD has remained below the threshold for a balanced population for all sample years. This indicates a population with few quality-size fish (Pope and Kruse 2007; Schramm and Willis 2012). With four years of natural reproduction in Deyo Reservoir, we would expect PSD values to be low, as it will take several more years to have a fully-developed population. However, maximum length of BG did not increase from 2016 to 2017. This suggests high natural and angler mortality is impacting the larger fish in this population, or they are not growing due to lack of food resources. The creel survey scheduled for 2019 should provide insight into angler mortality. Additionally, we recommend collecting spines to evaluate age, growth, and mortality of BG.

Some of the increase in mean BG length is due to poor recruitment since 2015, as none < 90 mm were sampled in 2017, and based on ages, no one or two year old fish were sampled. The high density of BG in the reservoir may be impacting their recruitment through cannibalism (Anderson and Weithman 1978; Guy and Willis 1991). Gear bias towards larger fish and variable recruitment could also be contributing factors (Hubert and Fabrizio 2007; Maceina and Pereira 2007). However, we do not believe gear bias is an issue, as BG 10 - 70 mm in length were sampled in Deyo Reservoir in both 2014 and 2015. Thus, the poor recruitment in Deyo Reservoir is likely a combination of high mortality from predation by LMB, cannibalism, and variable recruitment. If poor recruitment continues in future surveys, a more thorough evaluation will be needed to determine a management strategy for maintaining predator:prey balance.

Mean relative weight of Bluegill in 2017 was near 100 for all size classes. This is indicative of a population in “good” condition, and suggests that prey availability is not limited for different size classes (Pope and Kruse 2007; Willis et al. 2010). Even though density appears high, it is not a limiting factor for food resources.

Predator:prey balance

Both predator (LMB) and prey (BG) PSD in Deyo Reservoir were below the ranges considered to be indicative of balance. Based on the predator:prey decision model, this situation is generally caused by some combination of overharvest of LMB > 300 mm, high density of quality size LMB, overabundant BG due to low predation levels by LMB, and overharvest of larger BG (Schramm and Willis 2012). However, in Deyo Reservoir, LMB PSD is low because naturally recruited fish have just entered the stock size range and the only fish > 300 mm were those translocated previously in 2012 and 2016. This applies to BG as well, as they have not had time to grow into quality size. As such, the decision model should be used with caution as the fishery in Deyo Reservoir matures, and is more appropriate for established populations.

However, without angler mortality data, the extent of angler harvest influence on LMB PSD is unknown. Additionally, the lack of recruitment for both species is concerning. This scenario is consistent with an overabundant BG population that limits reproduction and survival of both species (Guy and Willis 1991; Schramm and Willis 2012). However, our data suggests that BG are not overabundant. We would expect PSD values should improve for both predator and prey over the next few years as the BG population increases in average size, and predation increases as more LMB reach >300 mm due to the translocated fish and restrictive harvest regulations. Further evaluation of predator:prey balance will determine if additional management options (translocation, modifying regulations, selective removal) are needed.

CONCLUSIONS

Preliminary analysis suggests that the restrictive regulations and translocation of LMB have been partially successful with improvement in overall size structure. However, the translocated LMB did not grow as expected to help fill in the gap in fish > 250 mm. The lack of recent recruitment in both LMB and BG is concerning, although the lack of BG recruitment could be a positive, as mortality will reduce the population and thus predation on LMB fry. This should improve LMB recruitment and therefore improve predator:prey balance over time. If recruitment of either species does not improve, additional supplementation will be necessary. However, with only one year of post-treatment data, additional surveys are needed to fully evaluate the impacts of the LMB translocation and regulation changes. Thus, we recommend conducting additional fish population surveys to fully evaluate the effectiveness of the LMB regulations and translocation on increasing the abundance, size structure, and recruitment of LMB, and improving the predator:prey balance of the reservoir. These surveys should include collection of spines and scales from LMB and BG for evaluating age, growth, and mortality. A creel survey should also be conducted to evaluate angler harvest of LMB and BG, whether BG are reaching desirable size for harvest, and angler satisfaction/awareness of the new restrictive regulations for LMB.

MANAGEMENT RECOMMENDATIONS

1. Continue current 406-mm minimum size limit, with a two fish bag limit for Largemouth Bass to reduce harvest of LMB, and to increase predation on small Bluegill.

2. Reassess fish population in 2019 to evaluate if the Largemouth Bass translocation and regulations have impacted the population’s size structure and recruitment, and the predator:prey balance of the reservoir.

3. Conduct angler surveys in 2019 to evaluate angler satisfaction, and angler harvest impact on the Largemouth Bass and Bluegill populations

4. In the future, aging marking (PIT tags) translocated fish would help evaluate the effectiveness of this management strategy.

Table 4. Comparisons of mean CPUE (fish/h), mean length (mm), PSD, and mean relative weight (W_r) among years (2014 - 2017) for Largemouth Bass collected by electrofishing Deyo Reservoir, Idaho. Significance was set at α = 0.10.

Year	CPUE	P-value	Mean length	P-value	PSD	Mean W_r	P-value
2014	53 (±17)	0.185	91 (±14)	< 0.001	33	89	0.235
2015	29 (±13)		200 (±25)		31	96	
2016	56 (±25)		214 (±20)		79	89	
2017	60 (±10)		264 (±18)		34	89	

Post-hoc Tukey-Kramer Test between years for LMB mean length		
Group 1	Group 2	P-value
2014	2015	< 0.001
2014	2016	< 0.001
2014	2017	< 0.001
2015	2016	0.858
2015	2017	0.003
2016	2017	0.005

Table 5. Comparisons of mean CPUE (fish/h), mean length (mm), PSD, and mean relative weight (W_r), among years (2014 - 2017) for Bluegill collected by electrofishing Deyo Reservoir, Idaho. Significance was set at α = 0.10.

Year	CPUE	P-value	Mean length (mm)	P-value	PSD	Mean W_r	P-value
2014	866 (±262)	0.200	76 (±1)	< 0.001	0.0	106	< 0.001
2015	1,331 (±557)		87 (±1)		0.1	135	
2016	809 (±169)		99 (±1)		1.4	99	
2017	724 (±255)		109 (±1)		1.7	100	

Post-hoc Tukey-Kramer Test between years			
		P-value	
Group 1	Group 2	Mean length	Mean W_r
2014	2015	< 0.001	0.998
2014	2016	< 0.001	< 0.001
2014	2017	< 0.001	< 0.001
2015	2016	< 0.001	< 0.001
2015	2017	< 0.001	< 0.001
2016	2017	< 0.001	0.623

Figure 13. Starting locations for electrofishing transects on Deyo Reservoir, Idaho, in 2014 - 2017.

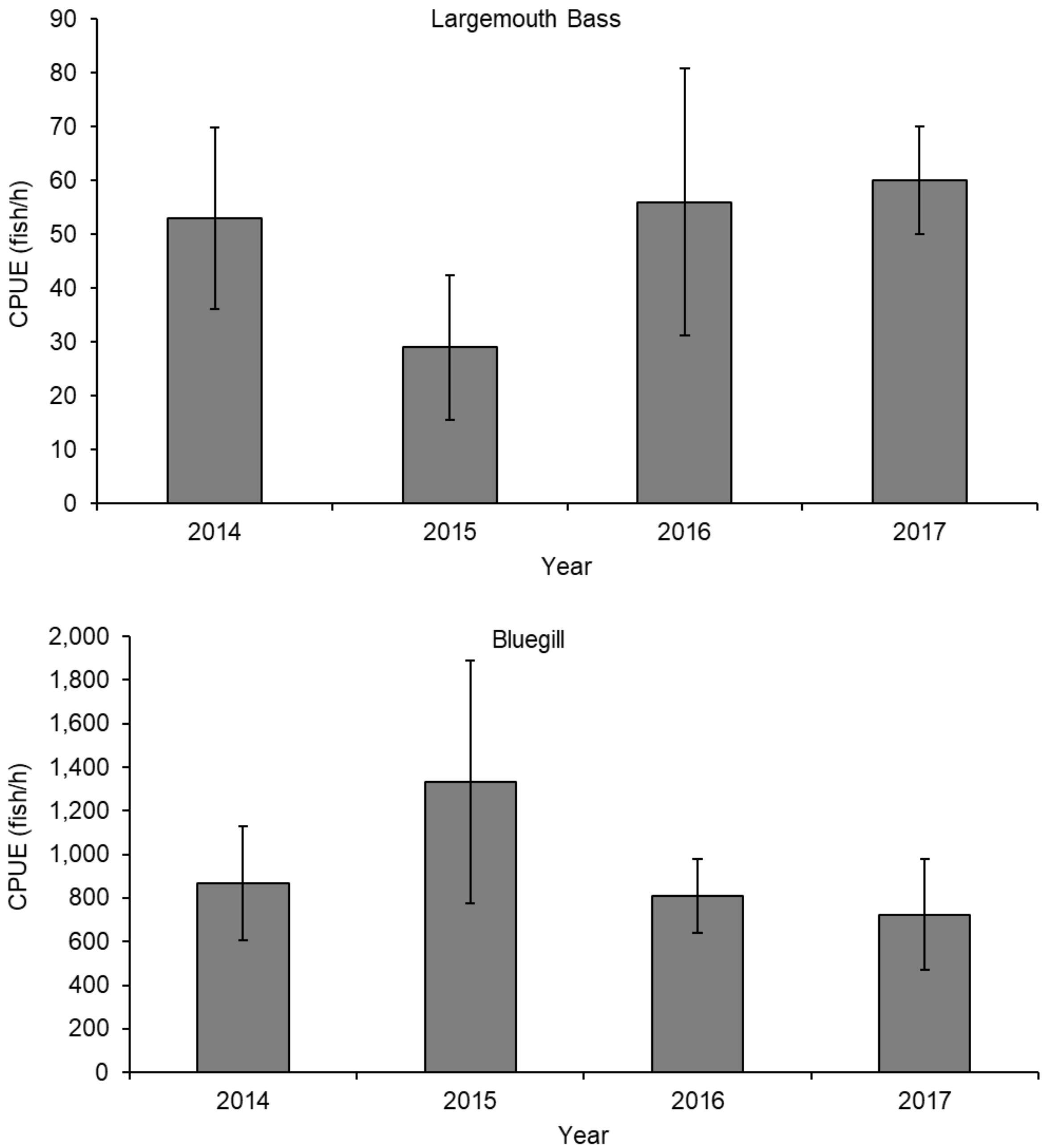

Figure 14. Catch-per-unit-effort (CPUE) of Bluegill and Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2017. Error bars represent 90% confidence intervals.

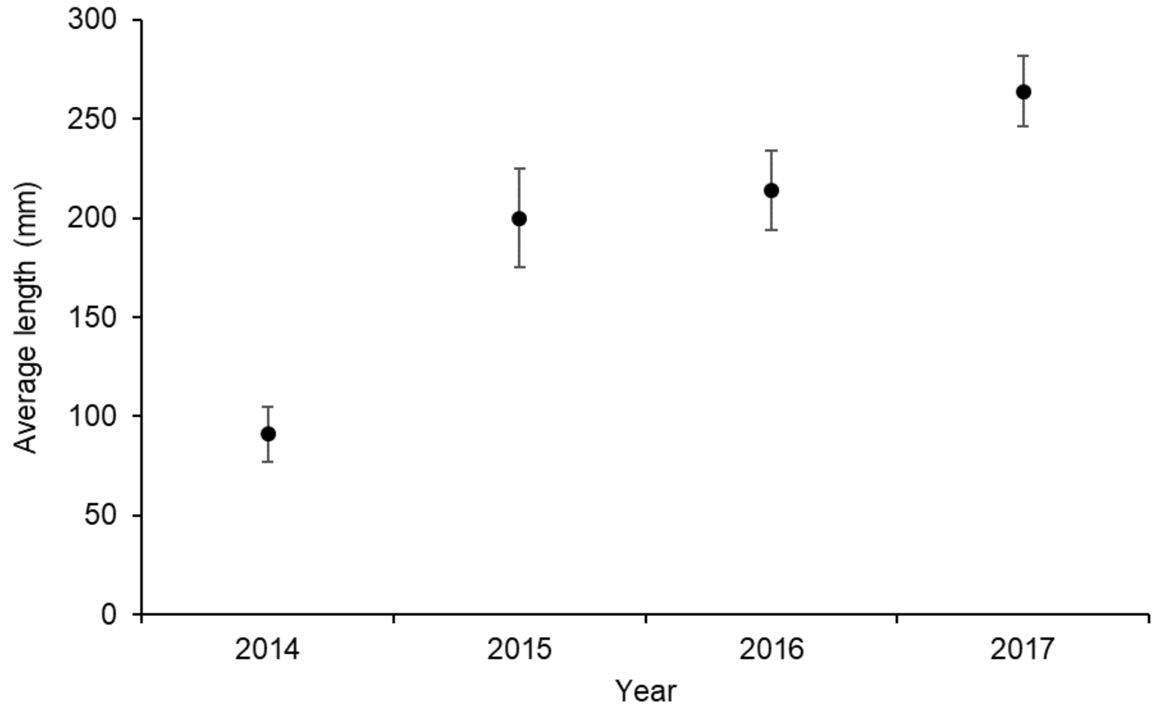

Figure 15. Average length of Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2017. Error bars represent 90% confidence intervals.

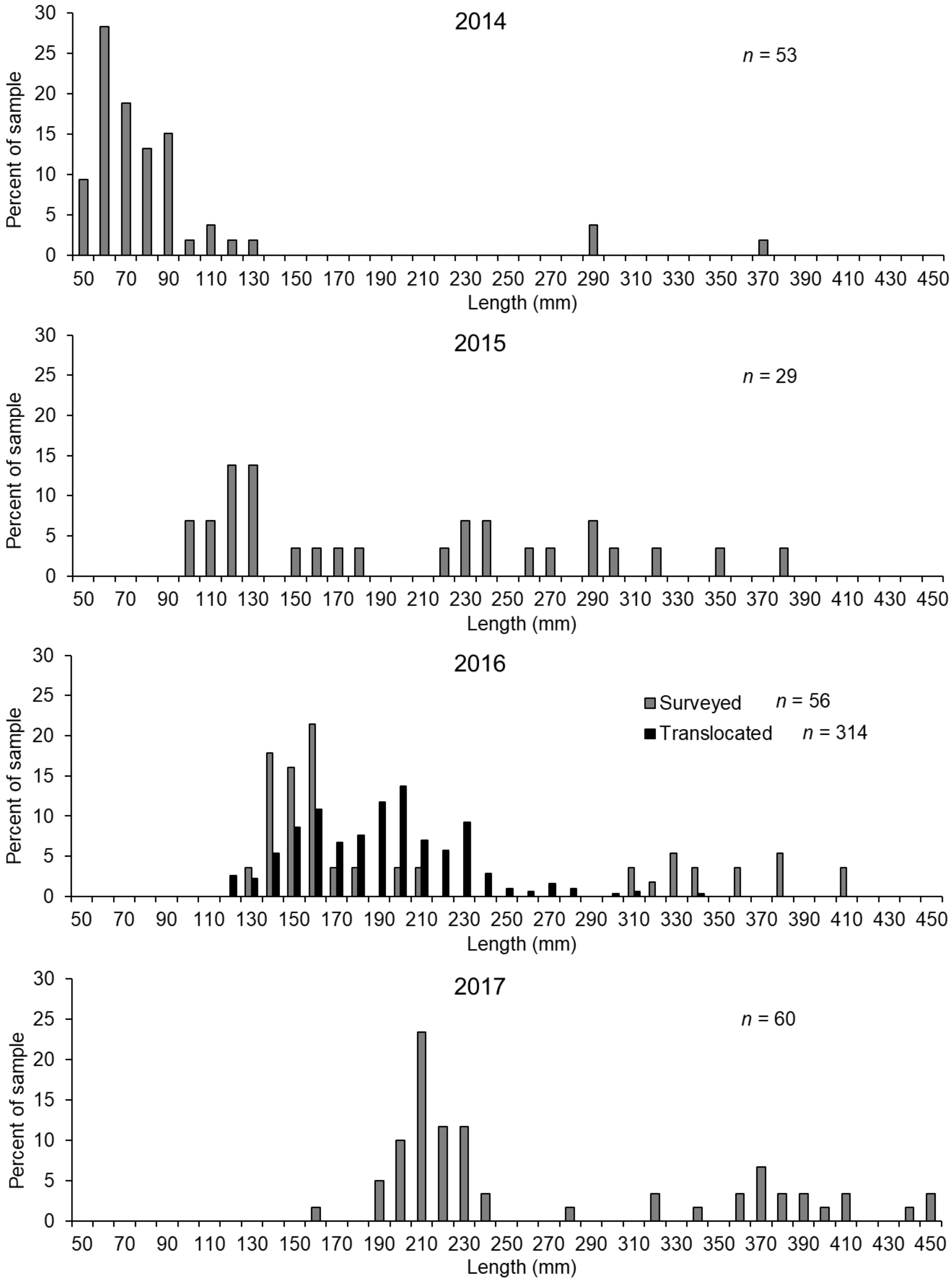

Figure 16. Length-frequency distribution of Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2017. The 2016 chart includes the length frequency of Largemouth Bass translocated into Deyo Reservoir prior to the survey.

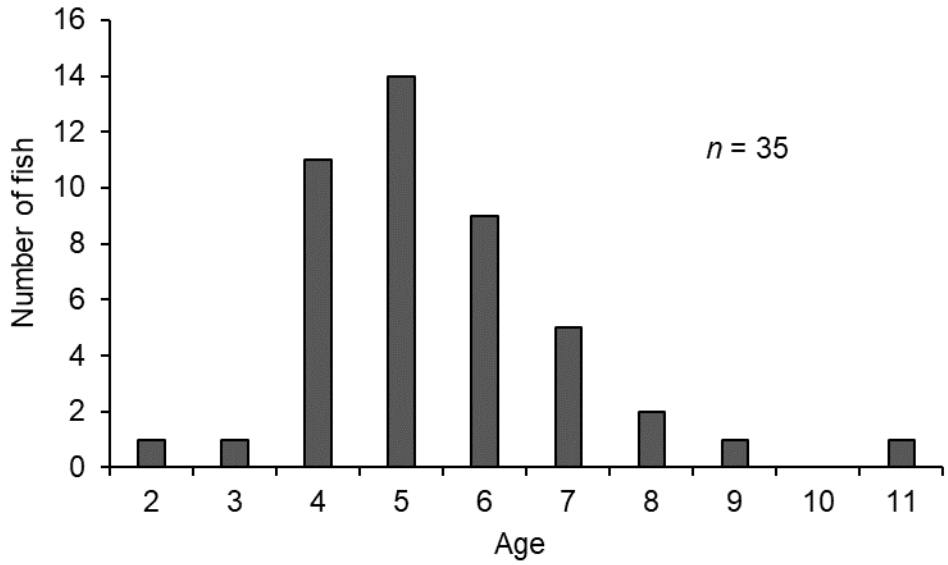

Figure 17. Age-frequency distribution of Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, in 2017, as estimated from scales.

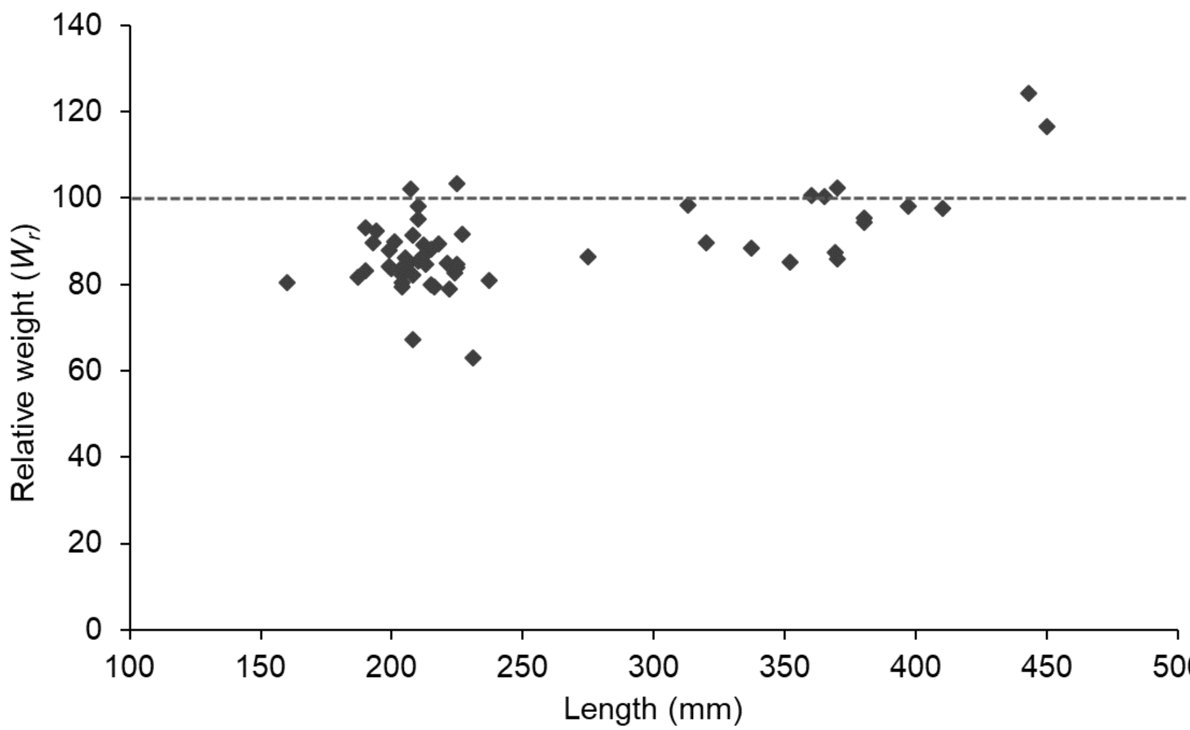

Figure 18. Relative weight values for individual Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, in 2017.

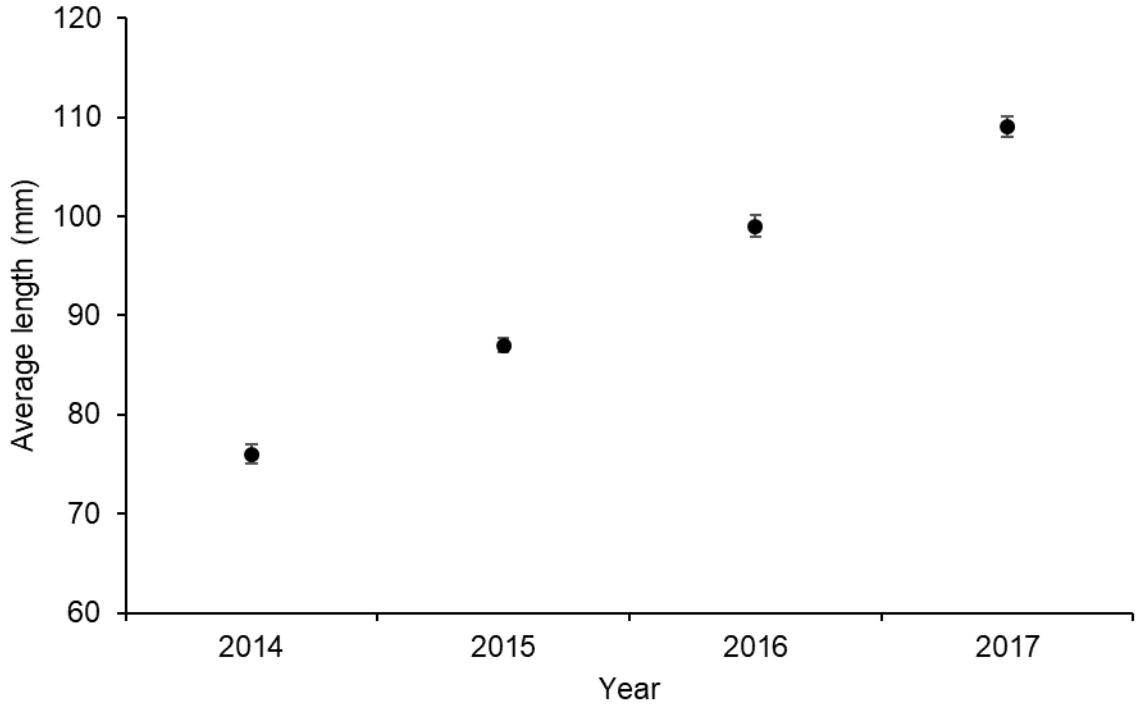

Figure 19. Average length of Bluegill sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2017. Error bars represent 90% confidence intervals.

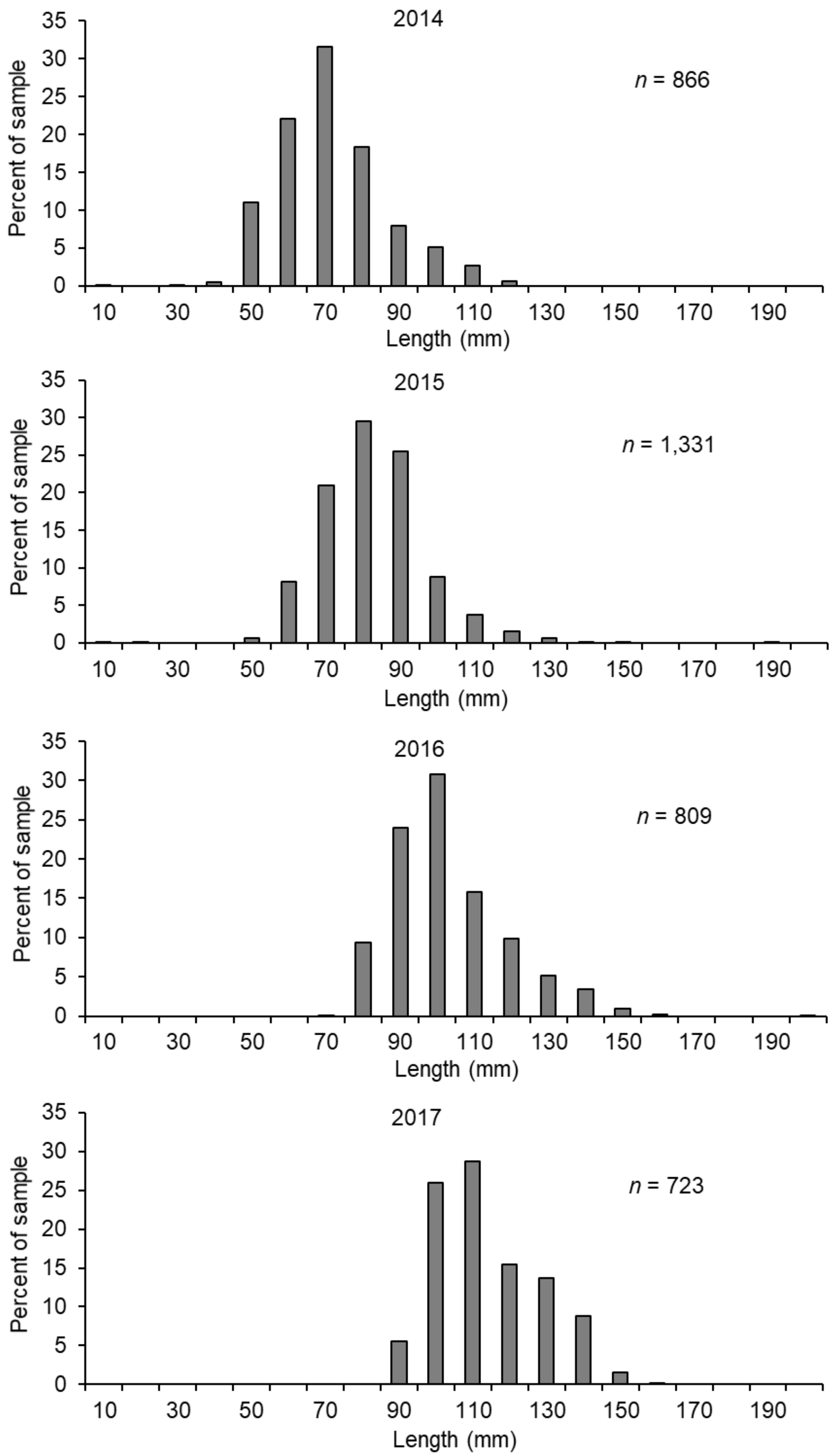

Figure 20. Length-frequency distribution of Bluegill sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2017.

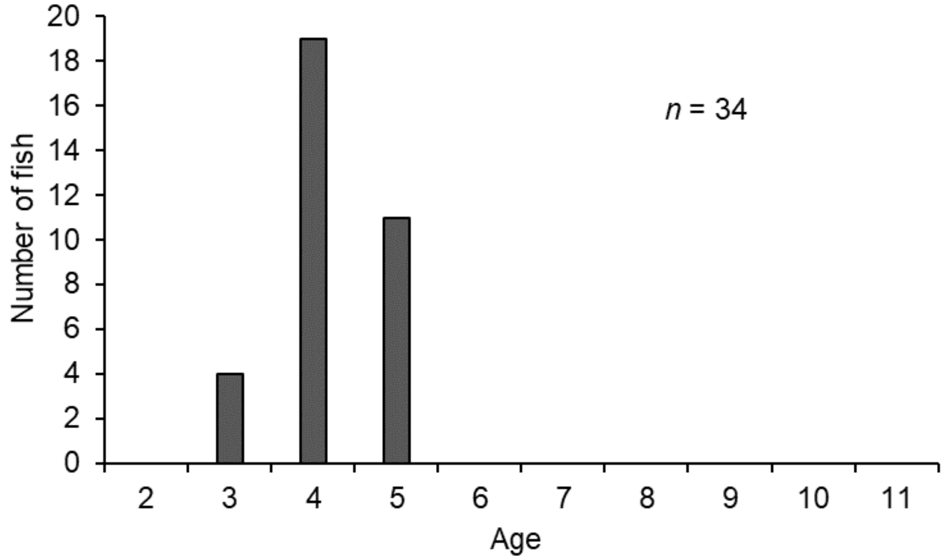

Figure 21. Age-frequency distribution of Bluegill sampled by electrofishing Deyo Reservoir, Idaho, in 2017, as estimated from scales.

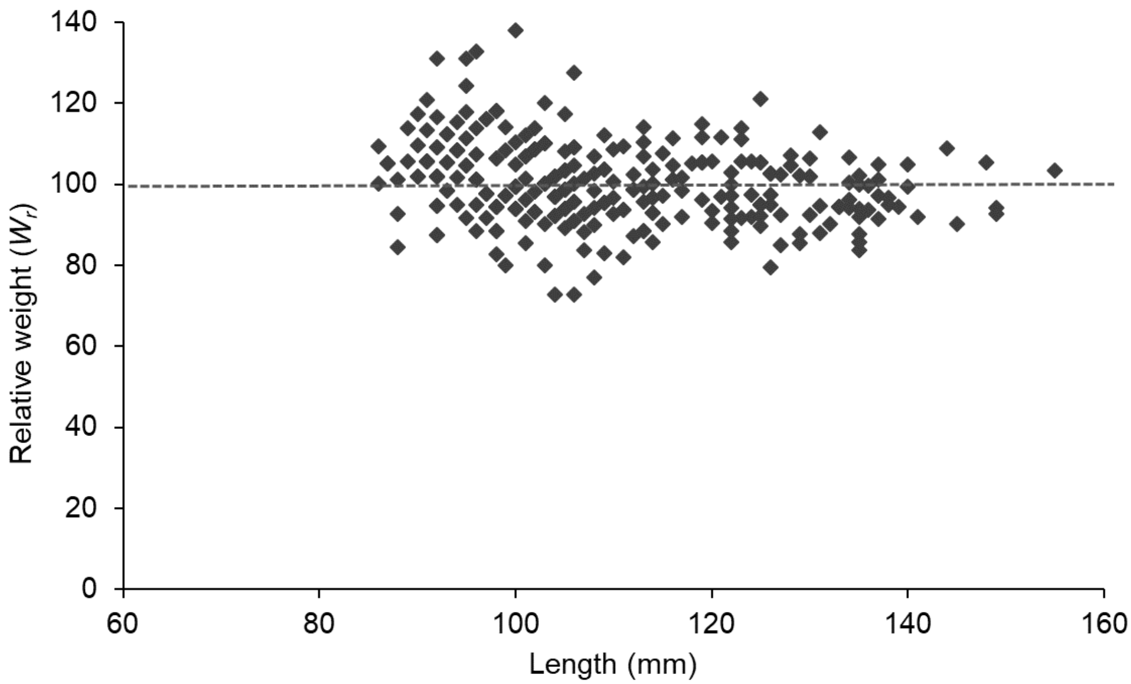

Figure 22. Relative weight values of individual Bluegill sampled by electrofishing Deyo Reservoir, Idaho, in 2017.

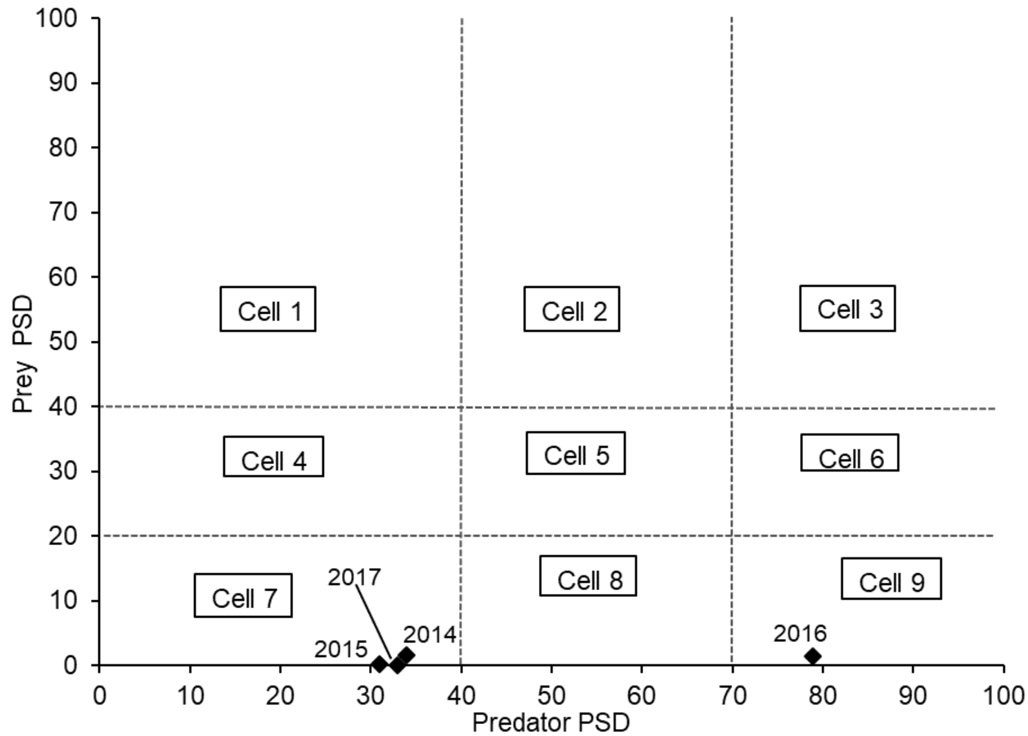

Figure 23. Comparison of predator (Largemouth Bass) and prey (Bluegill) proportional size distribution (PSD) collected through electrofishing Deyo Reservoir, Idaho, from 2014 to 2017. Dashed lines define the nine predator:prey PSD size structure possibilities based on Schramm and Willis (2012).

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DWORSHAK RESERVOIR FISHERY INVESTIGATIONS

ABSTRACT

Dworshak Reservoir provides a popular fishery for kokanee *Oncorhynchus nerka* (KOK) and Smallmouth Bass *Micropterus dolomieu* (SMB). Trends in angler effort, catch and harvest were monitored, along with annual use, exploitation, and abundance of SMB, to determine the efficacy of current fishing regulations. From March 22 to July 30, 2017, anglers fished an estimated 19,384 days or 109,916 h, caught 142,396 fish, and harvested 81,103, including 74,533 KOK, 4,313 SMB, and 2,257 other species. The catch rate for KOK anglers (1.3 fish/h) was near the median for 2014-2017, and the mean length of harvested KOK (mean = 269 mm TL) was the highest for this period. The catch rate for SMB (1.2 fish/h) was the lowest for 2014-2017 and the size of harvested bass (mean = 375 mm TL) was the highest for this period. Annual exploitation of SMB in 2017 was estimated at 11.4%, the lowest estimate to date. Exploitation in 2017 was lowest for stock size SMB, and similar for all other size classes. Exploitation of memorable and trophy SMB was higher in 2017 than in previous years. The abundance of SMB ≥ 200 mm TL was 37,087, which is similar to previous estimates. Indications are that the current management of these fisheries is sustainable, but the bass fishery should be monitored to ensure that the trophy component is maintained.

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INTRODUCTION

Dworshak Reservoir was the most popular fishing destination in Clearwater County and the second most popular destination in the Clearwater region, based on total angler trips in 2011 (Thomas MacArthur, IDFG, *unpublished data*). It provides a multi-species fishery for naturally reproducing kokanee *Oncorhynchus nerka* (KOK), Smallmouth Bass *Micropterus dolomieu* (SMB), and Westslope Cutthroat Trout *O. clarkii lewisi* (WCT), as well as hatchery-stocked Rainbow Trout *O. mykiss* (RBT). The reservoir also provides habitat for Bull Trout *Salvelinus confluentus* (BT), which are listed as Threatened under the Endangered Species Act.

A nutrient restoration program has been implemented annually since 2007 in cooperation with the U.S. Army Corps of Engineers. This program has increased both the productivity and efficiency of the food web by adding nitrogen (N) in the form of ammonium nitrate on a weekly basis from May through September. The addition of N has promoted the growth of edible phytoplankton instead of inedible and potentially toxic cyanobacteria, increased the biomass of *Daphnia*, an important prey source for planktivorous fish, and increased the growth of KOK at a given density (Wilson and Corsi 2015). Increasing the productivity of the KOK population is expected to improve the performance of both the KOK fishery, as well as fisheries for piscivorous fish that feed on KOK (i.e. SMB; Wilson and Corsi 2019).

With the exception of WCT and BT, there are no special fishing rules for Dworshak Reservoir. The daily bag limit for KOK is 25, with no minimum length. The daily limit for bass is six, with no minimum length. The daily limit for trout is six, with no minimum length. However, only two trout may be CT, none of which may be under 356 mm, 25 Brook Trout may be retained, and no harvest of BT. With the exception of SMB, these rules have received little, if any, scrutiny from the angling public. However, a number of bass anglers have expressed concerns that current regulations may result in a reduction of opportunity for trophy SMB fishing.

The fish management plan specifies that KOK fisheries should be managed for high yield, and specifies an average size ≥ 254 mm (10 inches) and catch rate > 1 fish/h. The plan further specifies both yield and quality opportunities for SMB within the Clearwater drainage. A creel survey was conducted in 2017 to assess the status and trends of this fishery and evaluate if it this fishery is meeting our management goals.

OBJECTIVES

1. Estimate angling effort, harvest, and catch rates for the Dworshak Reservoir fishery as a whole and for the two predominant target species, KOK and SMB.

2. Estimate annual use and exploitation of SMB.

3. Estimate the annual abundance of SMB ≥ 200 mm TL.

STUDY AREA

Dworshak Reservoir was impounded after the construction of Dworshak Dam in 1972 on the North Fork Clearwater River approximately 2.4 km from its confluence with the mainstem Clearwater River. The reservoir is narrow, steeply sloped, and primarily surrounded by coniferous forests. The North Fork Clearwater River and its tributaries drain nearly 632,000 ha, which is composed primarily of montane forests in steeply sloped terrain (Falter et al. 1977). The underlying geology is composed of Columbia River basalt and metamorphic sediments with granitic intrusions covered by shallow soils (Falter et al. 1977). Most of the North Fork Clearwater watershed above the reservoir lies within the Clearwater National Forest. The reservoir is immediately surrounded by land managed by the US Army Corps of Engineers (USACE), but much of the lower watershed is privately owned. Timber harvest is the primary commercial activity, although there is some agriculture in the lower watershed.

At full pool, Dworshak Reservoir is 86.3-km long with a surface area of 6,916 ha and a volume of 4.3 billion m^3 (Falter 1982). Typical annual drawdown lowers the pool elevation by 24 m and reduces the surface area by 27%. Peak pool elevation is typically reached by late June and drawdown begins after the first week of July, with typical minimum pool elevation reached by the second week of September. The mean hydraulic retention time is 10.2 months (Falter 1982) and the mean daily discharge from 2005-2016 was 152 m^3/s (http://www.cbr.washington.edu/dart/; accessed 4/11/20). Historically, Dworshak Reservoir begins to thermally stratify in April and stratification becomes pronounced from June through September. Destratification begins in the fall and occurs more rapidly at the upper end of the reservoir (Falter 1982; Wilson and Corsi 2016).

METHODS

A creel survey was conducted from March 22 through July 30, 2017. For this survey, we used an access-access design (Pollock et al. 1994). The survey was stratified by month and day type (weekday or weekend/holiday). Sampling days, shifts, and locations were chosen at random. Two days within each day type were selected for sampling each week with each day being given equal selection probability. The available daylight was divided into two shifts (am or pm) of equal length, and the selection probability was based on the relative number of interviews obtained during each time period from previous years. Access to the reservoir, whether by boat or shore, was limited to six locations (Figure 24). The two boat ramps at Bruce's Eddy were treated as separate access points and given independent selections probabilities, as were the ramp and marina at Big Eddy making, a total of eight possible access sites that could be chosen. Access points were assigned selection probabilities based on the relative number of interviews obtained from each during previous years and whether or not a ramp was usable at the time (ramp availability changed with pool elevation).

A single creel clerks was assigned to a shift at a given access point and instructed to remain at an assigned access point during the entire duration of the shift, or until all boat trailers and shore anglers were gone in the case of a pm shift. They were further instructed to make an effort to interview every party returning to the access site by boat, or departing from the access site by vehicle in the case of shore anglers. In the event that an interview could not be obtained, clerks recorded the party as unknown and noted the time of return. Lengths were collected from a random subsample of harvested fish. Multiple day trips, during which parties were camped at remote campsites, where treated as a single trip upon returning to the access site, and the hours fished and numbers of fish kept or released were recorded for the duration of the trip.

In an access-access survey design, data are only collected for completed trips, and total effort is estimated by expanding the effort documented for anglers returning to a given access point during a given shift by the probability of selecting that location and shift (Pollock et al. 1994). Daily effort ($\hat{e}_d$), measured in angler hours, was estimated in the following manner:

$$\hat{e}_d = \frac{e_{rsd}}{(\pi_r \times \pi_s \times \pi_b)}$$

Where: $\hat{e}_d$ = Estimated total fishing effort for day *d*.
e_{rsd} = Fishing effort sampled at site *r*, during shift *s*, on day *d*.
π_r = Selection probability of access site *r*.
π_s = Selection probability of shift *s*.
π_b = Probability of sampling a given boat during that shift.

The probability of sampling a given boat during a particular shift was simply calculated as the ratio of the number of boats sampled during that shift (including those that were not fishing) over the number returning (including those that were not sampled). Effort was also estimated in terms of angler days, which was calculated in a like manner. Angler days were calculated by summing the number of anglers fishing on a given day, irrespective of the time spent during the course of that day.

Total effort for a given strata was calculated by multiplying the mean daily effort for that strata by the number of days in the strata. Monthly effort was calculated by summing the effort of the strata within each month, and annual effort was calculated by summing the monthly effort.

Total catch and total harvest were estimated in the same manner as effort, substituting each into the above formulas. Formulas used to calculate standard errors for catch and effort can be found on pages 234-236 of Pollock et al. (1994). Catch rates were calculated by dividing total catch for the respective period by total effort. In addition, we calculated these metrics for anglers that targeted KOK or SMB only.

Estimates of effort, harvest, catch rate, and mean size of harvested fish were compared to estimates from 2014-2016 and confidence intervals were used to assess differences (Johnson et al 1999). Because the length of the survey has not been consistent between years, we only compared a period (April through July) that has been sampled consistently. This period encompasses nearly all of the effort directed toward KOK, but misses effort directed toward SMB during the late summer and fall.

Exploitation and use rates were estimated using the IDFG “Tag, You’re It” program. Exploitation was defined as the percentage of the population that was harvested in a given time period, and use was the percentage of the population that was caught, whether harvested or released, during that period. Smallmouth Bass were obtained from angling and at tournament weigh-ins, and tagged with a T-bar anchor tag. Exploitation and use were calculated following the methods of Meyer and Schill (2014), using the number of returns reported in the first 365 days after release. A reporting rate of 51.0% and tag loss rate of 0%, estimated in 2015 (Hand et al. 2020) were used to estimate exploitation and use. Exploitation and use were also calculated for the following size indices: stock = 180 to 279 mm TL, quality = 280 to 349 mm TL, preferred = 350 to 429 mm TL, memorable and trophy ≥ 430 mm TL (Gablehouse 1984; Nuemann et al. 2012). These estimates were compared to estimates from 2014-2016 and confidence intervals were used to assess differences.

The abundance of SMB ≥ 200 mm TL was estimated using a Lincoln estimator (Alisauskas et al. 2014). For this, we used the estimated harvest from the creel survey and the number of tags returned from SMB harvested during this same period. The unbiased estimator, as given by Alisauskas et al. (2014), is as follows:

$$\widehat{N} = \frac{(t+1)(\widehat{H}+1)\lambda}{(r+1)} - 1$$

In this equation, $\widehat{N}$ is the estimated abundance, Ĥ is the estimated harvest, t is the number of tags at large during the creel survey, r is the number of tags reported during the same period (March 22 through July 30) as the creel survey, and λ is the reporting rate. The variance for this estimate was calculated as follows:

$$var(\widehat{N}) = \left(\frac{t\widehat{H}}{r}\right) \times var(\lambda) + \lambda^2 \times var\left(\frac{t\widehat{H}}{r}\right)$$

Greater detail for calculating the variance can be found in Alisauskas et al. (2014).

The abundance estimate for 2017 was compared to estimates for 2016 and 2004 from Hand et al. (2020) and confidence intervals were used to assess differences.

RESULTS

There were 717 interviews collected during 64 creel shifts for Dworshak Reservoir between April 1 and July 30, 2017. From these, we estimated anglers fished 19,384 days (SE = 4,788 or 25%) or 109,916 h (SE = 25,183 or 23%; Table 6). Fishing effort peaked in June with 8,404 angler days (SE 3,765 = or 45%) and 43,138 angler h (SE = 16,848 or 39%; Table 6). Excluding March, which was a partial month, we documented the least effort in April with 1,472 angler days (SE = 516 or 35%) and 8,215 angler h (SE = 2,819 or 34%). The mean length of a single day of fishing was 5.7 h with a range of 0.1-15.1 h. Fishing party sizes ranged from 1 to 8 anglers with a mean party size of 2.2. Anglers caught 142,396 fish (SE = 31,909 or 22%), of which 81,102 (SE = 23,759 or 29%) were harvested. This included 77,986 KOK, of which 74,533 (96%) were harvested, 59,469 SMB, of which 4,313 (7%) were harvested, and 4,808 other species, of which 2,129 (44%) were harvested. Other species included RBT (4,270 caught, 2,018 harvested), BT (414 caught, no harvest), Black Crappie *Pomoxis nigromaculatus* (111 caught and harvested), and WCT (13 caught, no harvest).

Anglers specifically targeting KOK fished 9,932 days (SE = 2,618 or 26%) or 57,495 h (SE = 14,997 or 26%). Kokanee fishing effort was relatively low in April, increased through June, and then declined again in July (Table 7). Effort directed toward KOK from April through July of 2017 was near the median for 2014-2017 (Figure 25). Kokanee anglers caught an estimated 76,901 KOK (SE = 21,640 or 28%) and harvested 73,605 (SE = 20,136 or 29%), with a mean catch rate of 1.3 fish/h. Catch rates for KOK were lowest in April (mean = 0.1 fish/h), increased to a high in June (mean = 1.6 fish/h), then declined in July (mean = 1.2 fish/h). The combined harvest from April through July was near the median and the second lowest harvest rate for 2014-2017 (Figure 26). Most KOK anglers (81%) caught at least one KOK during a given day, 78% harvested a kokanee, and 9% harvested a limit of 25 KOK. The mean length of harvested KOK was 269 mm TL (Figure 27), which was greater than the previous three years (Figure 28). Kokanee anglers also caught an estimated 3,585 incidental species (SE = 1,610 or 45%), all of which were RBT or SMB, and harvested 1,604 of these (SE = 1,007 or 63%).

Anglers specifically targeting SMB fished 7,975 days (SE = 1,398 or 18%) or 46,266 h (SE = 8,959 or 19%). Bass fishing effort peaked in May, and declined through July (Table 7). Effort directed toward bass from April through July of 2017 was near the median for the past four years

(Figure 25). Bass anglers caught an estimated 56,936 SMB (SE = 12,756 or 22%) for a mean catch rate of 1.2 fish/h, and harvested 3,881 (SE = 887 or 23%). Catch rates for SMB were lowest in March (mean = 0.1 fish/h) and April (mean = 0.3 fish/h), and were similar from May through July (mean = 1.3 fish/h). Harvest in 2017 was the second lowest of the past four years, and harvest rate was the lowest during this period (Figure 29). Most bass anglers (73%) caught at least one bass during a given day, 20% harvested at least one bass, and 3% harvested a limit of six bass. The mean length of harvested bass was 375 mm TL (Figure 27), which was greater than 2014 and 2015, but similar to 2016 (Figure 28). Bass anglers also caught an estimated 1,621 incidental species (SE = 481 or 30%), and harvested 471 of these (SE = 229 or 49%).

A total of 235 SMB were tagged to evaluate exploitation and use in 2017. This included 63 stock size (180 to 279 mm TL), 41 quality size (280 to 349 mm TL), 72 preferred size (350 to 429 mm TL), and 59 memorable or trophy size (≥ 430 mm TL) SMB. Of these, 42 were returned by anglers within 365 days at large. Of these, 13 tags were returned for SMB that were harvested. All of these were caught from March through July, with 85% of harvested tag returns coinciding with the creel survey. Tags returned from fish that were released, or only harvested because of the tag, were caught from March through November, with 83% of the tag returns coinciding with the creel survey. The exploitation rate for all size classes was 11.4%, whereas use was 36.8%. Estimates of exploitation have tended to trend downward since 2013, although confidence intervals overlap for most years (Figure 30). Estimates of use were similar from 2013 through 2017.

Annual exploitation for 2017 was zero for stock size SMB, and similar for the larger size classes (Figure 30). Although no tags were returned for stock size SMB that anglers would have harvested if not tagged, there were harvested stock size SMB encountered in the creel. Annual exploitation for both preferred and quality SMB were lower in 2017 than the mean of the previous three years, whereas annual exploitation of memorable and trophy (combined) SMB was higher. Annual use of SMB tended to decline with increasing size class in 2017, but confidence intervals overlapped (Figure 31). The mean annual use for the previous three years was lowest for stock, memorable and trophy class SMB, and highest for quality and preferred size classes.

The abundance of SMB ≥ 200 mm TL was estimated at 37,087, with a 90% confidence interval from 26,709 to 47,466. Although this estimate was lower than 2016 (42,000) and 2004 (45,000), confidence intervals for all three estimates overlapped.

DISCUSSION

The KOK fishery on Dworshak Reservoir during 2017 was consistent with a declining abundance of adult KOK. The April through July timeframe was used to evaluate the KOK fishery because it typically encompasses the majority of the effort directed toward KOK. In 1990, 86% of KOK angling effort for the entire year was documented during this period (Maiolie et al. 1993). In 2015, 79% of the KOK angling effort from March through October occurred during this period (Wilson and Corsi 2016). When comparing trends from 2015 through 2017, angler effort, harvest, and CPUE all peaked in 2016 and then declined, whereas the mean length of harvested KOK continued to increase from a low observed in 2015. This fluctuation is typical of Dworshak Reservoir (Wilson and Corsi 2016), and common in other KOK systems throughout North Idaho (Reiman and Meyers 1992). While the density of adult KOK was observed to decline from 2015 through 2017, it did not decline below the range that has been observed since the population recovered from the 1997 flooding event (Wilson and Corsi 2019). Therefore, this fishery appears to be undergoing typical fluctuations and current regulations are sufficient to maintain it.

The mean catch rate and mean length of harvested KOK were above the management objectives of 1 fish/h and 254 mm (10 inches) TL during 2017. Catch rates for Dworshak Reservoir were similar to those reported for Lake Pend Oreille, a lake in northern Idaho, but much higher than those reported for reservoirs in southern Idaho, including Anderson Ranch Reservoir, Arrowrock Reservoir, and Lucky Peak Reservoir (Klein et al. 2020). Both Dworshak Reservoir and Lake Pend Oreille are characterized by higher population densities and smaller KOK. The reservoirs in southern Idaho are characterized by lower population densities and larger KOK, and reported catch rates in southern Idaho were < 0.5 fish/h (Klein et al. 2020).

Both prior creel survey and tag return data suggest that the April through July creel survey encompasses the majority of angling effort directed toward SMB. In 2015, 73% of the angling effort directed toward SMB from March through October occurred during this period. This was the longest duration for a creel survey in which SMB angling effort was tracked separately. However, 83% of tags reported by anglers during the entire year were returned during this period, suggesting that it represents the bulk of the SMB angling effort. This is particularly true for harvest. Although tags were reported throughout the year for SMB that were released, none were reported for SMB harvested after July.

The SMB fishery on Dworshak has also undergone fluctuations over the past four years, albeit on a different cycle. Angler effort, harvest, and CPUE for the SMB fishery all peaked in 2015, a low year for these metrics for the KOK fishery, and have been in decline since. However, like the KOK fishery, mean length has been increasing since 2014. These fluctuations and the underlying mechanisms have not been as well documented and understood as the KOK fishery. Further investigations, including the possible effects of KOK abundance, a potential food source for larger SMB, are warranted.

The overall exploitation of SMB in Dworshak Reservoir has shown a declining trend since 2007 and was low in 2017 compared to other fisheries throughout Idaho. The 2017 estimate (11.4%) was similar to the lowest estimate (11.8%, Cascade Reservoir) reported by Meyer and Schill (2014). While down from 2007, use has remained fairly constant over the past five years, suggesting that the decline in exploitation is likely due to an increase in the proportion of SMB that are released by anglers, rather than a decrease in the number caught. The overall low and declining trend in exploitation suggests that current harvest regulations are adequate for providing harvest opportunity without threatening the trophy component of the SMB fishery.

Patterns of exploitation by size class were different in 2017 from previous years. While we know from creel surveys that some exploitation of stock size SMB occurred, none was detected from tag returns. This is not strictly an artifact of sample size, as estimates of use for this size class were similar to other size classes. Therefore, it is likely that anglers released most stock size SMB they caught. In other years, not only has exploitation calculated from tag returns been higher for this size class, but it has been more prevalent in angler creels as well. Since the mean length of harvested SMB was higher in 2017 than recent years, it is possible that anglers simply released nearly all of the stock size SMB they caught in favor of harvesting larger fish. In years where exploitation was higher for stock-sized SMB, harvest oriented anglers likely did not encounter as many larger SMB and kept stock-sized SMB instead. Most notable was the apparent increase in exploitation of memorable- and trophy-size SMB during 2017. However, caution should be used in interpreting these results. Black bass obtained from anglers, as in the present study, may be more likely to be caught again, thus resulting in higher estimates of exploitation (see Table 10 in Butts et al. 2016). Therefore, estimates of exploitation presented here may be biased high. Monitoring should be continued to determine if the recent increase in exploitation of memorable- and trophy-size SMB is an anomaly or a trend. If exploitation is found to remain significantly higher for these size categories, an analysis of how it affects the trophy component of the fishery may be warranted.

Estimates of SMB abundance suggest that the population has been relatively stable since 2004. The estimate of abundance for 2017 was similar to estimates for 2016 and 2004 (Hand et al. 2020). This provides additional evidence that current regulations are sufficient to maintain the SMB fishery.

MANAGEMENT RECOMMENDATIONS

1. Maintain current fishing regulations for Dworshak Reservoir.

2. Continue monitoring annual exploitation of SMB.

3. Investigate effects of KOK abundance on SMB growth and population trends.

Table 6. Angler effort (reported as angler days and hours), catch and harvest estimated from a creel survey of Dworshak Reservoir from March 22 to July 30, 2017. Estimates are reported for all anglers combined.

	Angler		Kokanee		Bass		Other	
	Days	Hours	Caught	Harvest	Caught	Harvest	Caught	Harvest
March	284	1,432	338	333	33	17	133	128
April	1,472	8,215	252	236	1,261	341	952	482
May	5,385	35,728	21,248	18,863	24,188	1,749	1826	208
June	8,404	43,138	40,300	40,026	24,557	1,552	1746	1314
July	3,839	21,403	15,848	15,075	9,429	654	285	125
Totals	19,384	109,916	77,986	74,533	59,468	4,313	4,942	2,257

Table 7. Angler effort (reported as angler days and hours), catch, CPUE (fish/h), harvest and mean total length (TL) of harvested fish estimated from a creel survey of Dworshak Reservoir from March 22 to July 30, 2017. Estimates are reported separately for anglers who only targeted kokanee or Smallmouth Bass.

	Angler		Target species				Incidental species	
	Days	Hours	Caught	CPUE	Harvest	TL	Caught	Harvest
			Kokanee anglers					
March	229	1,176	338	0.3	333	277	122	117
April	363	2,094	224	0.1	208	267	12	12
May	2,454	16,223	20,996	1.3	18,621	266	1,435	200
June	4,875	24,975	39,819	1.6	39,692	265	1,959	1,257
July	2,011	13,027	15,524	1.2	14,751	271	57	18
Total	9,932	57,495	76,901	1.3	73,605	269	3,585	1,604
			Smallmouth Bass anglers					
March	39	197	22	0.1	6	390	22	22
April	827	4,720	1,210	0.3	329	410	589	178
May	2,834	18,579	23,535	1.3	1,721	330	548	0
June	2,858	15,921	23,262	1.5	1,190	433	396	205
July	1,417	6,849	8,905	1.3	636	340	65	65
Total	7,975	46,266	56,934	1.2	3,882	375	1,620	470

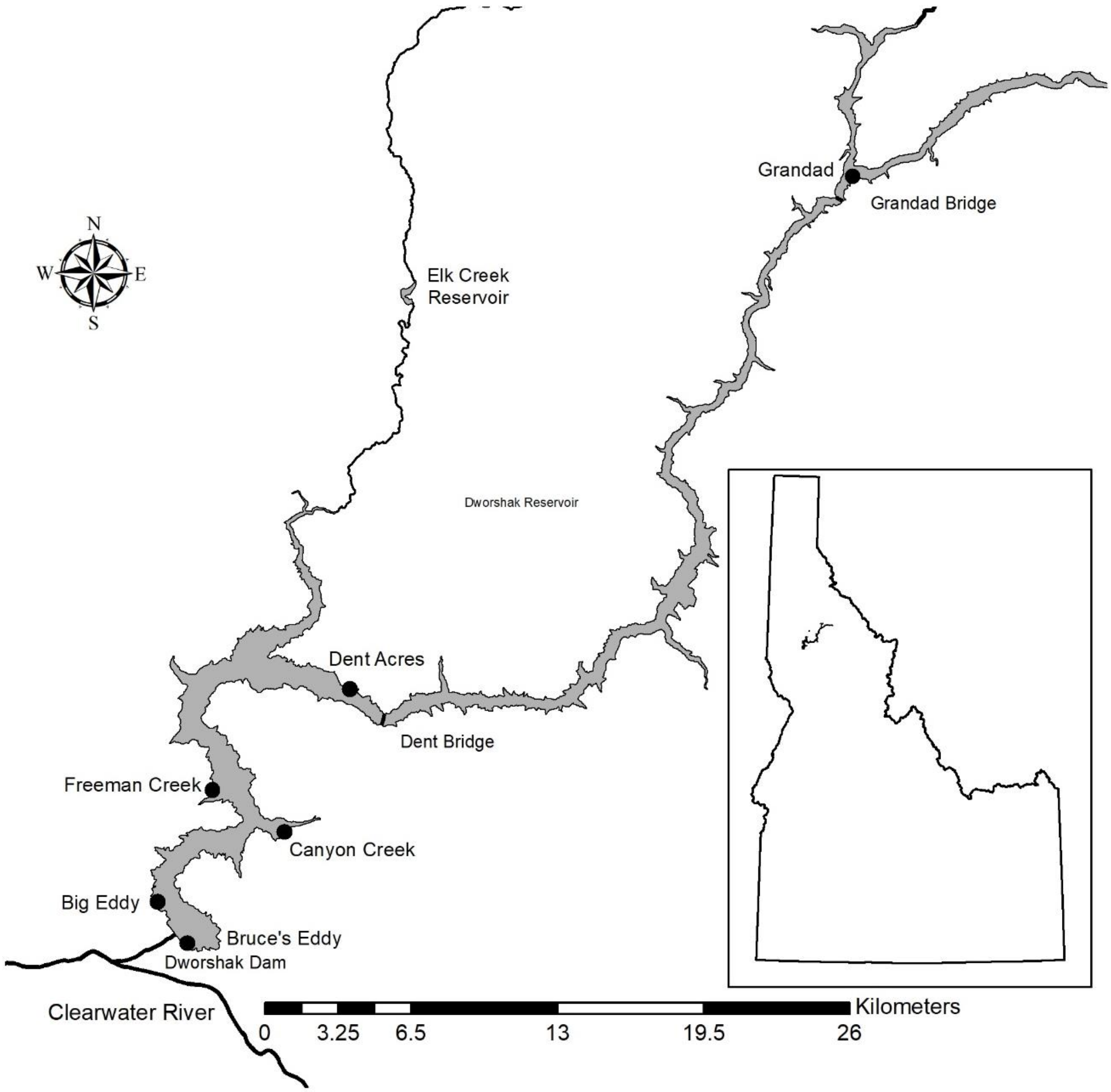

Figure 24. Fishing and boating access sites for Dworshak Reservoir, Idaho where creel surveys were performed in 2017.

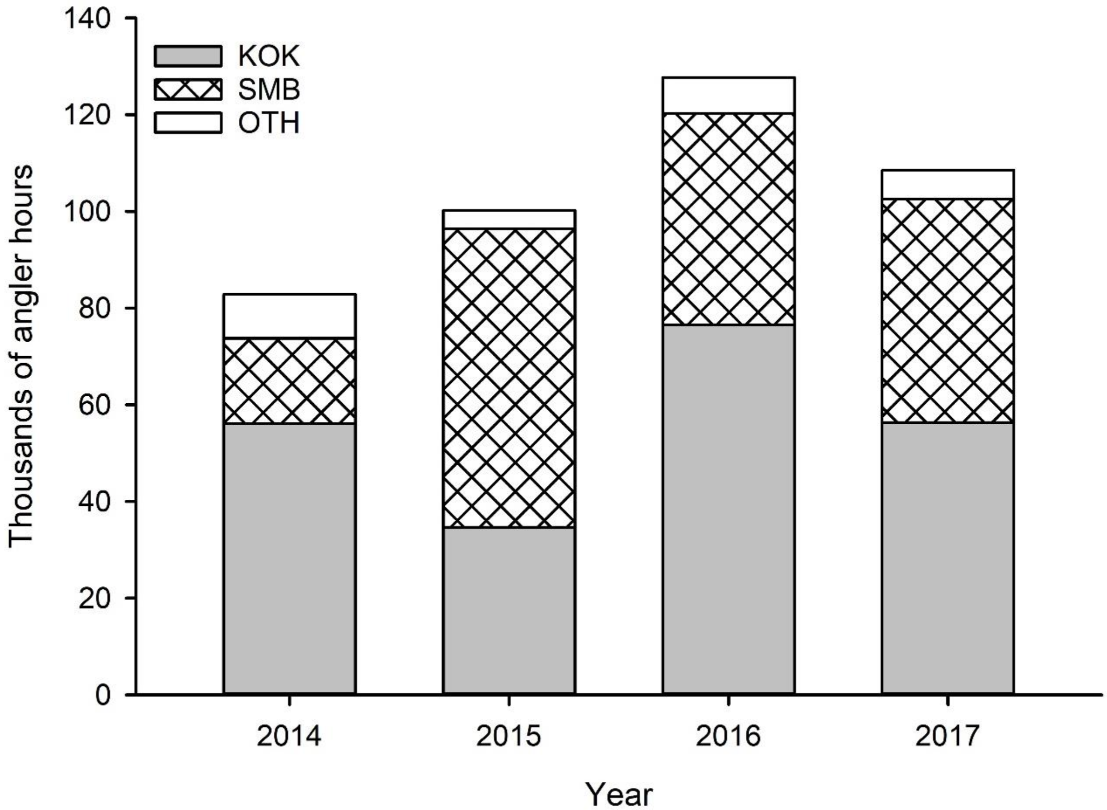

Figure 25. Fishing effort on Dworshak Reservoir from April through July of 2014 through 2017. Effort is reported by those targeting only kokanee (KOK), only Smallmouth Bass (SMB), and all other fishing effort (OTH).

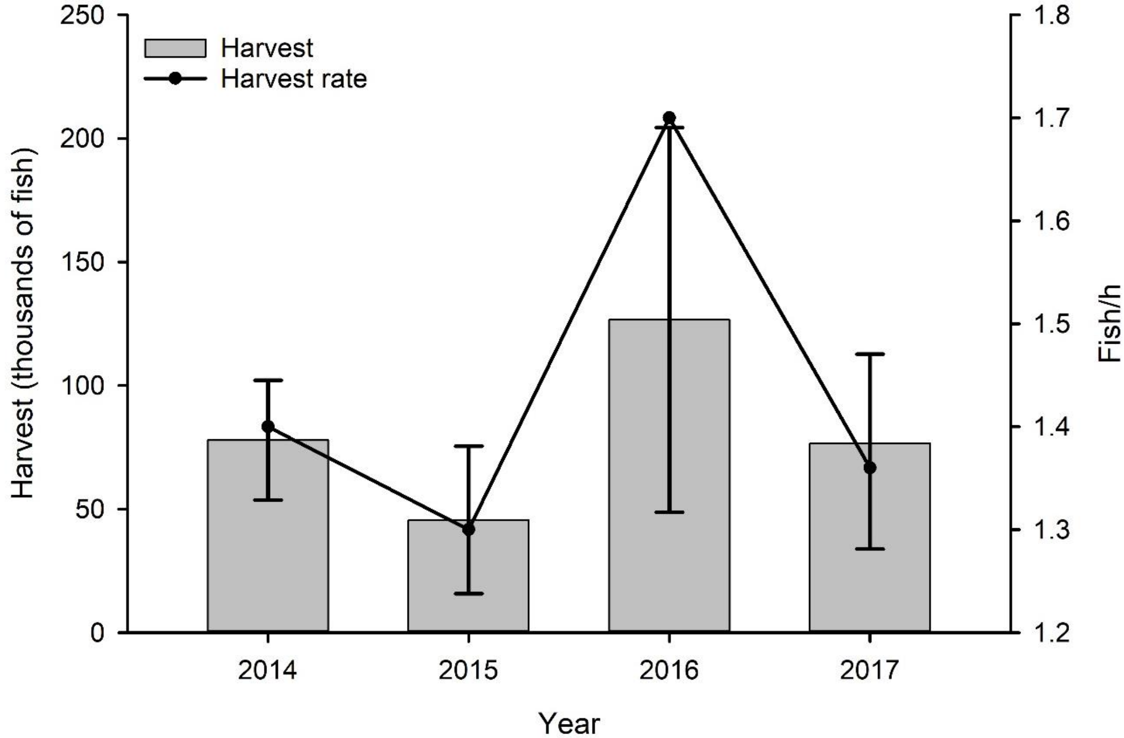

Figure 26. Harvest and harvest rates (fish/h) of kokanee by anglers targeting kokanee, as determined through creel surveys on Dworshak Reservoir from April through July of 2014 through 2017. Error bars represent 95% confidence intervals for harvest.

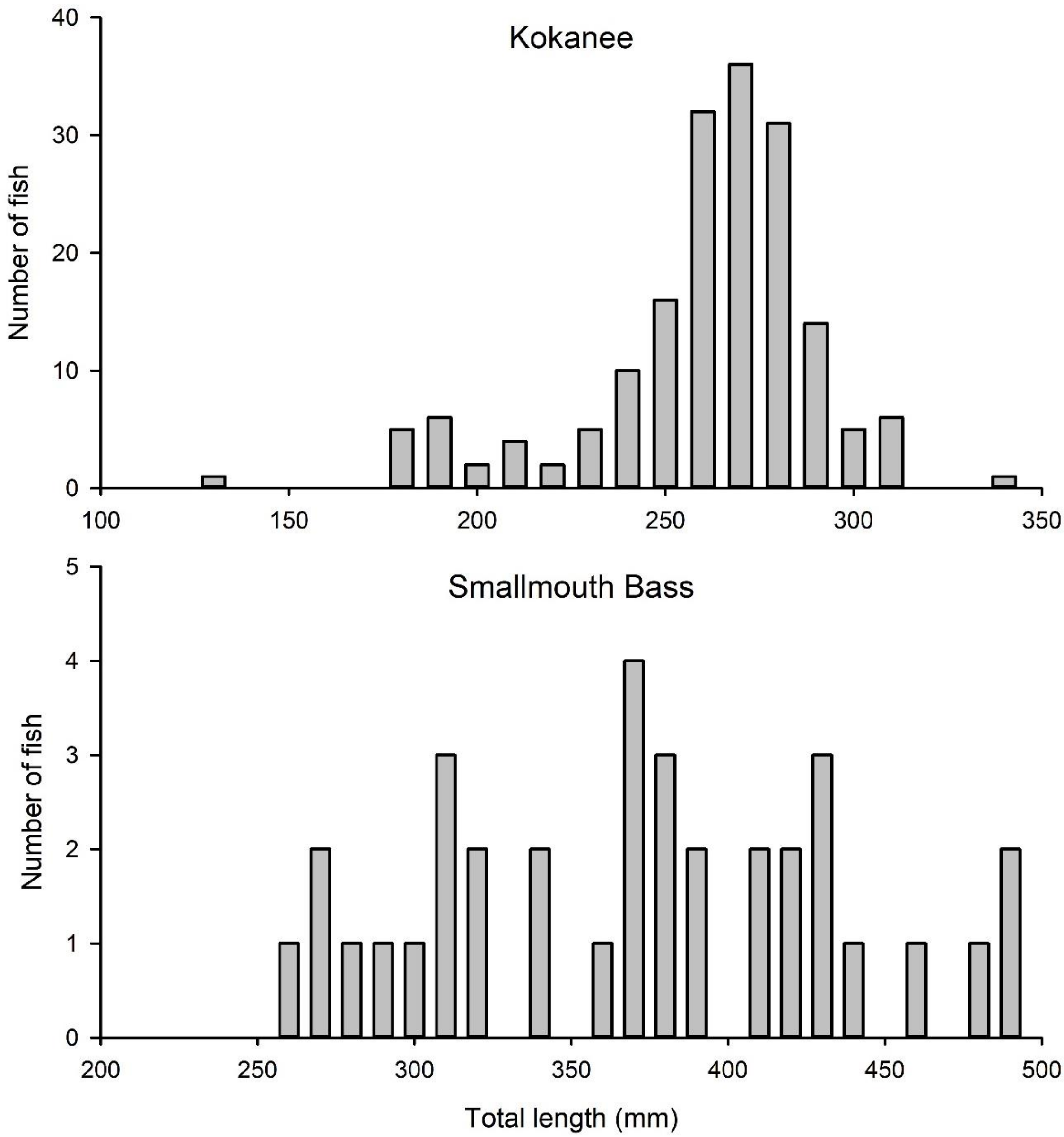

Figure 27. Length-frequency distributions for 176 kokanee and 35 Smallmouth Bass measured in a creel survey at Dworshak Reservoir from April through July of 2017.

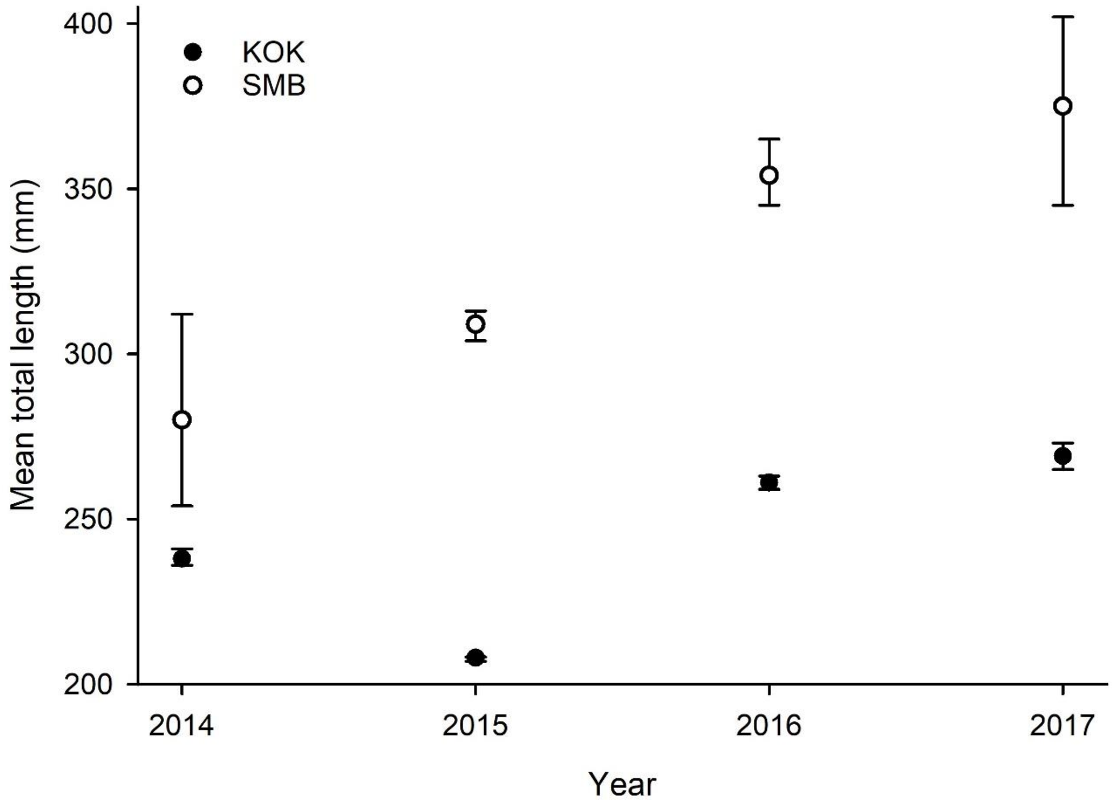

Figure 28. Mean length of kokanee (KOK) and Smallmouth Bass (SMB) harvested from Dworshak Reservoir as determined from creel surveys conducted April through July of 2014 through 2017. Error bars represent 95% confidence intervals.

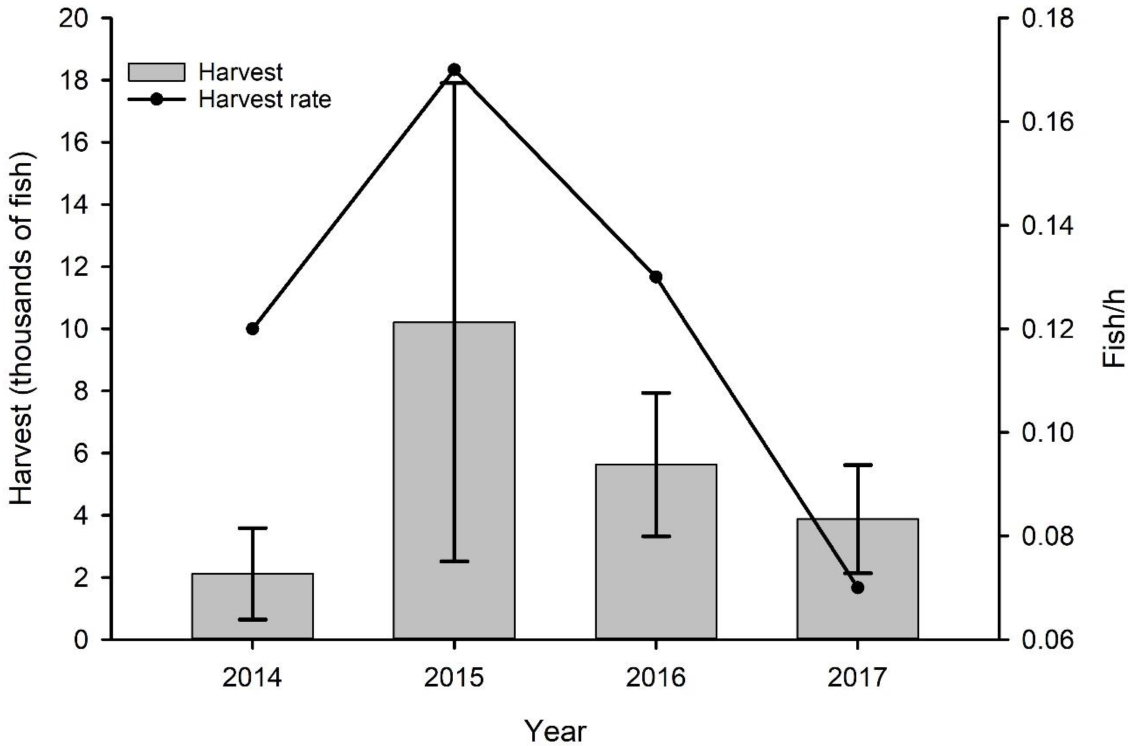

Figure 29. Harvest and harvest rates (fish/h) of Smallmouth Bass by anglers targeting bass as determined from creel surveys on Dworshak Reservoir from April through July of 2014 through 2017. Error bars represent 95% confidence intervals for harvest.

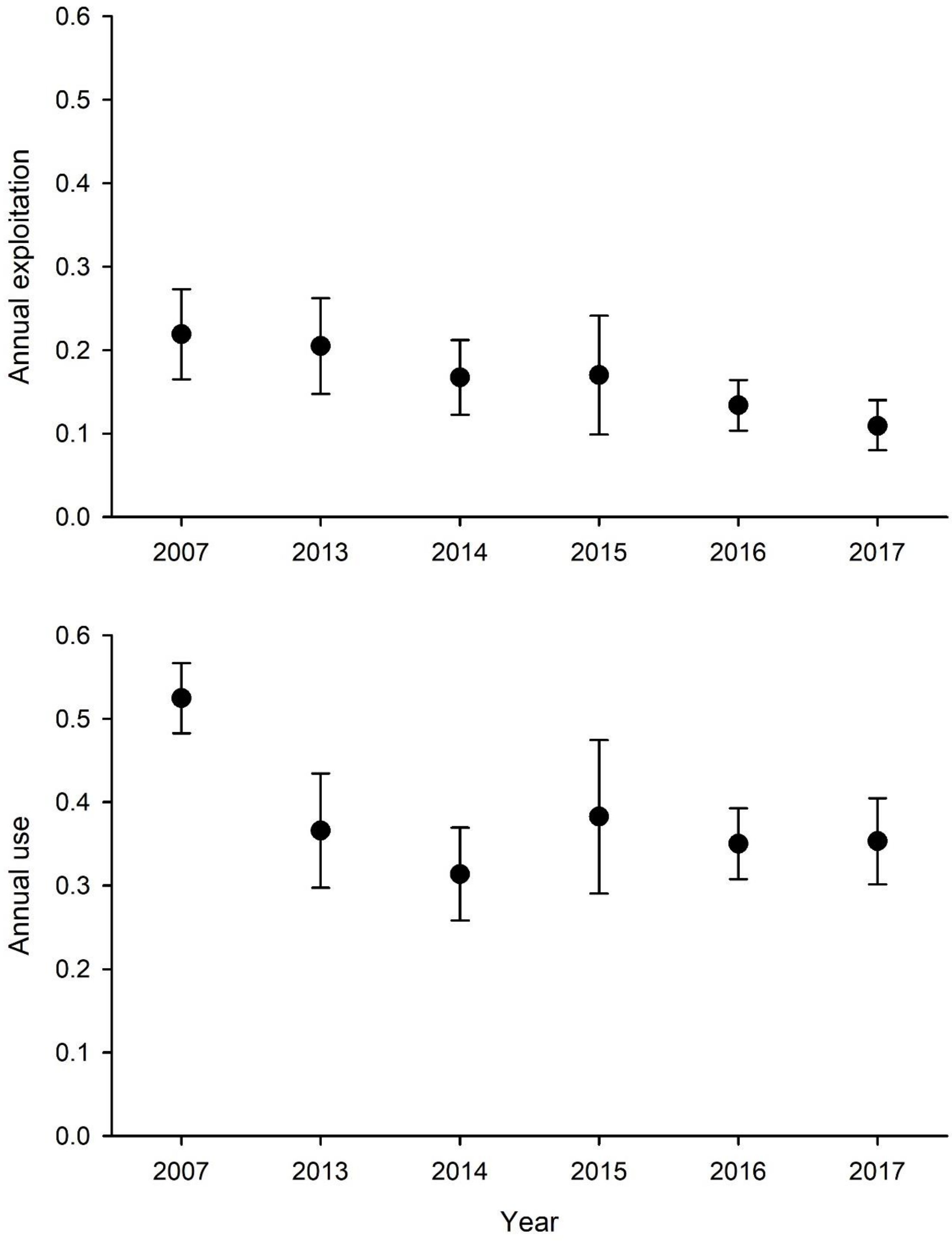

Figure 30. Annual exploitation and use rates for Smallmouth Bass caught from Dworshak Reservoir as estimated from tags returned by anglers. Error bars represent 90% confidence intervals.

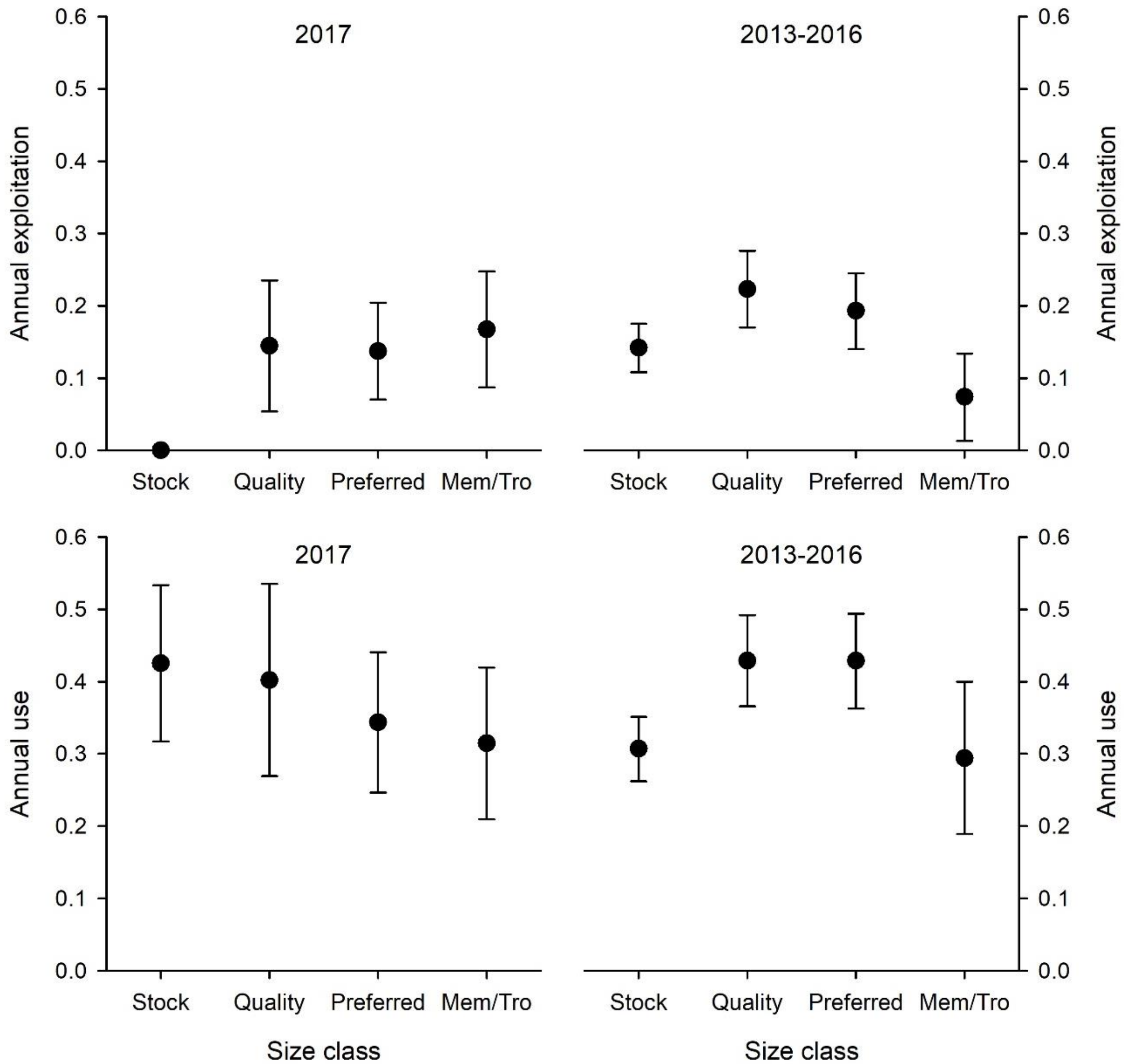

Figure 31. Annual exploitation and use rates for Smallmouth Bass caught from Dworshak Reservoir by four length classes; stock (180-279 mm TL), quality (280-349 mm TL), preferred (350-429 mm TL), and both memorable and trophy (Mem/Tro, ≥ 430 mm TL). The top graphs show exploitation and the bottom show use. The graphs on the left show estimates for 2017, and the right show mean values for 2013-2016. Rates were estimated from tags returned by anglers and error bars show 90% confidence intervals.

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EVALUATION OF TROUT POPULATIONS IN THE LOCHSA RIVER

ABSTRACT

A snorkel survey was conducted in the Lochsa River drainage in 2017 to assess trends in Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT), Rainbow Trout *O. mykiss* (RBT), and Mountain Whitefish *Prosopium williamsoni* (MWF) density and size distribution. Mean density for all species declined from 2013 to 2017. Additionally, RBT density in 2013 and 2017 was < 2% of historic surveys. Larger size (> 305 mm) WCT were more abundant in upper surveys sections, while few larger RBT were observed in the Lochsa River drainage. The recent declines in density of these species has been observed in other north Idaho rivers, and has been attributed to low discharge and higher temperatures. Discharge did not appear to be an issue in the Lochsa River drainage, but lower than normal winter temperatures and higher than normal summer temperatures were recorded during this time period. This could have reduced fish density in the Lochsa River through increased mortality. Alternatively, fishes may have migrated upstream out of index sites to seek colder water. For RBT, specifically, density may have been affected by cessation of hatchery stockings. Prior to 1990, ~11,000 RBT were stocked in the Lochsa River annually, while none have been stocked since. This, combined with recent declines in steelhead smolt out-migration, and adult returns throughout the Clearwater River basin, are other likely contributing reasons for reduced RBT density compared to historic surveys. While our data in the Lochsa River is short-term, declines in MWF densities have been observed in the main-stem of other northern Idaho rivers. While the direct cause of these declines has not been identified, these declines have been linked to occurrences of low flows and higher water temperatures. Smallmouth Bass have not been observed in surveys of the Lochsa River. However, future surveys should continue to evaluate the distribution of these non-native fish, as they may experience a climate-mediated spread throughout the upper Clearwater River system

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT) are distributed throughout the Lochsa River drainage, occupying the main-stem river and tributaries. Both resident and fluvial life history forms are present. US Highway 12, which runs along the Lochsa River, was completed in 1962. Its completion opened up the entire length of the Lochsa River to easy access for anglers. By 1966, the WCT population was considered to have been drastically reduced, likely due to high levels of harvest (Mallet 1967; Dunn 1968; Rankel 1971; Lindland 1977). A 1956 creel survey estimated WCT catch at 5,948 fish (Corning 1956). By 1976, creel surveys showed catch had dwindled to 654 WCT (Lindland 1977). The decline of this WCT population prompted the implementation of catch-and-release regulations upstream of the Wilderness Gateway Campground bridge in 1977.

The Lochsa River watershed also supports wild runs of spring and summer Chinook Salmon *O. tshawytscha*, summer steelhead, and Pacific Lamprey *Entosphenus tridentatus*. Additionally, hatchery releases of summer Chinook salmon occur in this watershed. Resident Rainbow Trout, Bull Trout *Salvelinus confluentus* (BT), and Mountain Whitefish *Prosopium williamsoni* (MWF) also occur in the watershed. Bull Trout are located mainly in the upper main-stem Lochsa River and the higher elevation streams, and MWF occur primarily in the main-stem river and the largest tributaries. Currently, the management strategy for resident fish in the Lochsa River basin is to maintain a high abundance larger WCT and RBT, promote MWF fishing, and maintain a catch-and-release BT fishery (IDFG 2019).

Since 1975, snorkel surveys have been conducted in the Lochsa River to monitor the WCT population. Densities increased seven-fold in the catch-and-release section, and four-fold in the harvest section from 1977 to 1981 after the catch-and-release regulations were implemented (Lindland 1982). Although occasional snorkeling surveys have been conducted in the Lochsa River since 1981, the survey in 2013 marked the first occasion since 1981 where the trend surveys established by Graham (1977) were revisited, thus allowing for a direct comparison of observed densities across time (Hand et al. 2008). The primary objective of the 2013 survey was to re-establish trend monitoring to evaluate current WCT densities, while simultaneously establishing trend and presence/absence surveys for other resident fishes, especially Smallmouth Bass (SMB) *Micropterus dolomieu*, which were believed to have migrated into the lower Lochsa and Selway rivers.

The Lochsa River provides a popular trout fishery that is often compared to other premier Idaho fisheries. While there is some harvest opportunity in the Lochsa River system, it is managed with restrictive or catch-and-release regulations. In 2017, the Lochsa River drainage had three harvest management strategies for trout to provide a diversity of opportunity for anglers and to maintain a high density of larger WCT. For the main-stem Locsha River from the mouth to the Wilderness Gateway Bridge, the daily limit was two trout none under 356 mm from Memorial Day weekend through November 30. For the main-stem Lochsa River from the Wilderness Gateway Bridge (WGB) to the confluence of Colt Killed and Crooked Fork Creeks and Crooked Fork from its mouth to Brushy Fork, the trout season and limit was catch and release, open all year. Crooked Fork Creek upstream of Brushy Fork Creek and all other tributaries of the Lochsa River were managed under the Clearwater Region general rules (harvest of two trout, any size, open all year). As demand on these fisheries continues, it is important to track the status of fish populations to ensure continued quality fishing and to conserve wild native trout populations.

OBJECTIVES

1. Assess whether a high density of larger Westslope Cutthroat Trout is being maintained in the Lochsa River.

2. Evaluate whether the density and size structure of Rainbow Trout *Oncorhynchus mykiss* (resident rainbow trout and juvenile steelhead) and Mountain Whitefish *Prosopium williamsoni* in the Lochsa River are stable or improving.

3. Evaluate whether the distribution of Smallmouth Bass is increasing in the Lochsa River.

STUDY AREA

The headwaters of the Lochsa River drainage are found in the Bitterroot Mountains on the Idaho-Montana border (Figure 32). The Lochsa River is formed by the confluence of Crooked Fork Creek and Colt Killed Creek (formerly White Sands Creek) and flows 113 km southwest, joining the Selway River at the town of Lowell, ID, to form the Middle Fork Clearwater River. The Lochsa River drainage covers 3,056 km^2, all in Idaho County. The majority of the watershed occurs at elevations > 1,200 m. Most of the sub-basin is granitic rock that is part of the Idaho batholith. Land ownership in the Lochsa River drainage is mixed, with the majority of the land under public ownership managed by the U.S. Forest Service. Nearly 80% of the drainage is designated as wilderness (Selway Bitteroot Wilderness Area) or roadless. The Lochsa River is designated a Wild and Scenic River. The primary private landowner in the drainage is Western Pacific Timber Company. They, and previous owners, have intensively managed this area for timber production. These actions are believed to impact fish populations in some areas through sedimentation, poor in-stream cover, and impacts from upland disturbances.

METHODS

Field sampling

A snorkel survey was conducted in the Lochsa River basin from August 12 to 15, 2017 to monitor the density and size structure of WCT, Rainbow Trout/steelhead (RBT), and MWF. We surveyed a total of 38 transects in the main-stem Lochsa River, Crooked Fork Creek and Colt Killed Creek (Figure 32). Detailed transect descriptions and locations are provided in Appendix A of Hand et al. (2016). Snorkel surveys were conducted by one or two snorkelers, depending on the width of each transect. A single snorkeler was used only when the entire wetted width of the stream could be effectively observed by one person. The number of snorkelers surveying each transect was consistent with previous surveys to allow for direct comparison of data. Transects were snorkeled downstream, with each surveyor swimming close enough to shore to see the shoreline. Each snorkeler surveyed towards the thalweg and towards their respective shorelines. All fish observed were counted, and length was estimated to the nearest inch for most species. Other species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson 50 m reel style fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. This distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead and resident Rainbow Trout are indistinguishable, and are collectively referred to as “RBT”.

DATA ANALYSIS

We evaluated trends in WCT and RBT density following methods from historic surveys in the Lochsa River (1975 - 1981) which calculated linear density as fish/100 m (Graham 1977; Mabbott 1980; Lindland and Pettit 1981). Those surveys doubled the transect length to account for when two snorkelers were used. Main-stem Lochsa River survey sections were delineated as follows: mouth of Lochsa River to Fish Creek, Fish Creek to Lake Creek, and Lake Creek to Crooked Fork Creek. Mean density was calculated by averaging transect densities to maintain consistency with other Idaho Department of Fish and Game snorkel projects (Putnam et al. 2018). We evaluated long-term trends in mean density using least squares regression with survey year (1975 - 2017) as the independent variable and $\log_e$ transformed mean density as the dependent variable (Maxell 1999; Kennedy and Meyer 2015). The intrinsic rate of change in the population (r_{intr}) is determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance. The trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$.

We also evaluated trends in WCT, RBT, and MWF density, and density of just those fish > 305 mm, using area measurements to be consistent with modern estimates for snorkel surveys conducted in Idaho and allow for comparisons to other rivers (Apperson et al. 2015). These areal densities were calculated as “fish/100 m^2”, and were compared to the previous survey conducted in 2013 as “higher” or “lower” due to only having two years of data. Main-stem Lochsa River survey sections were delineated as follows: mouth of Lochsa River to WGB, WGB to Lake Creek, and Lake Creek to Crooked Fork Creek. Density was also compared between years for Crooked Fork Creek and Colt Killed Creek. Mean density was calculated by averaging individual transect densities to maintain consistency with other Idaho Department of Fish and Game snorkel projects (Apperson et al. 2015). A more thorough statistical analysis of population trends using area density will be conducted after additional surveys have been completed. Distribution and densities of WCT, RBT, and MWF were visually represented by plotting densities observed at each transect on maps of the survey area using GIS software.

RESULTS

Westslope Cutthroat Trout

Westslope Cutthroat Trout had the highest linear density observed in the main-stem Lochsa River (Table 8). They were observed in 89% of transects, with linear density the lowest in the Mouth to WGB survey section. In 2017, mean linear density of WCT for all main-stem Lochsa River transects was the lowest since 1979 and about 45% of what was observed in 2013 (Figure 33). The largest declines in mean linear density occurred in the two upper-most survey sections (Table 8). The proportion of WCT observed > 305 mm in length for the main-stem Lochsa River was 47%, similar to 2013 (49%; Table 9). In 2017, the proportion of larger WCT observed declined in the two lower survey sections compared to 2013. The Lake Creek to Crooked Fork Creek survey section was the only location where the proportion of larger WCT increased from 2013 to 2017. Although mean linear density has been higher since 1980, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in density for WCT in the main-stem Lochsa River (Table 10).

Mean areal density of WCT in the Lochsa River drainage was the lowest for any species (Table 11). They were observed in 92% of transects, with the highest areal densities occurring in Crooked Fork Creek (Figure 34). In 2017, mean areal density of WCT was about 25% of what was observed in 2013 (Figure 35). The largest declines occurred in the WGB to Lake Creek and Colt Killed Creek survey sections (Figure 35). The proportion of WCT observed > 305 mm in length for the Lochsa River drainage was 43%, lower than 2013 (50%; Table 9). In 2017, mean areal density of WCT >305 mm was lower in all management sections compared to 2013 (Figure 36). The largest decline in proportion of larger WCT observed occurred in Crooked Fork Creek.

Rainbow Trout

Rainbow Trout had the lowest linear density observed in the main-stem Lochsa River (Table 8). They were observed in 46% of transects, with linear density the lowest in the Mouth to WGB survey section. In 2017, mean linear density of RBT for all main-stem Lochsa River transects was the lowest of any survey conducted since 1975. In fact, linear density in both 2013 and 2017 was < 2% of any previous survey (Figure 33). The largest declines in mean linear density occurred in the Mouth to WGB survey section (Table 8). In 2017, no RBT > 305 mm in length were observed in the Lochsa River drainage, a decline from 2013 (3%). There was a statistically significant decline ($r_{intr} < 0$) in linear density of RBT from 1975 to 2017 (Table 10). This was due to the decline in density in 2013 and 2017.

Mean areal density of RBT in the Lochsa River drainage was the highest for any species (Table 11). They were observed in 74% of transects, with the highest areal densities occurring in Crooked Fork Creek (Figure 37). In 2017, mean areal density of RBT was over two times higher than what was observed in 2013 (Figure 38). This overall increase resulted from the increase in density within Crooked Fork Creek, as areal densities declined in all other survey sections (Figure 38).

Mountain Whitefish

Mean areal density of MWF in the Lochsa River drainage was 0.22/100 m^2 (Table 11). They were observed in 82% of transects, with the highest areal densities occurring in Crooked Fork Creek (Figure 39). In 2017, mean areal density of MWF in the Lochsa River drainage was < 25% of what was observed in 2013 (Figure 40). The largest declines occurred in the WGB to Lake Creek and Lake Creek to Crooked Fork Creek survey sections (Figure 40). In contrast, mean areal density increased in Colt Killed Creek. In 2017, mean areal density of MWF >305 mm was lower in all harvest management sections compared to 2013 (Figure 41). The proportion of MWF observed > 356 mm in length for the Lochsa River drainage was 63%, higher than 2013 (51%). In 2017, mean areal density of MWF >305 mm was lower in all management sections compared to 2013 (Figure 41). The largest declines in proportion of larger MWF observed occurred in the WGB to Lake Creek and Lake Creek to Crooked Fork Creek survey sections.

Smallmouth Bass

No Smallmouth Bass were observed in 2013 or 2017 in the Lochsa River drainage.

DISCUSSION

Westslope Cutthroat Trout

From 1975 to 2017, a significant trend in WCT linear density was not detected in the main-stem Lochsa River. In contrast, increasing trends in WCT density has been observed in the Selway, North Fork Clearwater (NFCR), North Fork Coeur d'Alene (CDAR), and St. Joe (SJR) rivers (*see* Selway River chapter in this report; Hand et al. 2020; Ryan et al. 2020). We must caution that our data series for the main-stem Lochsa River contains a 30-year gap (1982-2012) and therefore may not properly represent trends in density. Additionally, the long-term trend would be increasing had density not declined in 2017. Despite the lack of a trend, density was still at least seven times higher in 2013 and 2017 than in surveys conducted prior to 1978. Additional surveys will allow for a more thorough analysis of population trends in the future.

Mean density of all sizes of WCT, and WCT > 305 mm, declined from 2013 to 2017 in all four harvest management zones. Declines in density of all sizes of WCT and WCT >305 were also observed in the NFCR, South Fork Clearwater River (SFCR), and SJR during this time frame as well (*see* SFCR chapter in this report; Hand et al. 2020; Ryan et al. 2020). This may represent actual declines in density or could due to other factors such as movement of fish out of areas snorkeled, observer variability, or decreased detectability (Sloat et al. 2005; Copeland and Meyer 2011). Because this decline in WCT density was observed in multiple rivers with different seasons and limits stretched across northern Idaho, it suggests the decline in density was driven by environmental conditions that either increased natural mortality or resulted in fewer fish being observed (fish moved to areas not snorkeled or were more difficult to see).

Low flow and higher temperatures were considered to be primary causes of observed declines in the CDAR and SJR (Ryan et al. 2020). Lochsa River discharge during sampling was similar for 2013 (17.0 m^3/s) and 2017 (15.6 m^3/s; USGS 2021), suggesting that it was not likely an issue for detectability. Discharge has been shown to have a positive correlation with trout survival on a 3 - 4 year time lag (Copeland and Meyers 2013). Mean annual discharge for the Lochsa River was below the 30-year average for three of the four years preceding 2017, but only one of the four years preceding 2013. The negative impacts of low flow on trout survival include redd scour, displacement of fry, and reduced overwinter survival (Jager et al. 1999; Carline and McCollugh 2003; Zorn and Nuhfer 2007). Thus, discharge may have had a negative impact on survival in the years prior to sampling in 2017.

Severe winter conditions are known to negatively impact salmonid populations in smaller streams through impacts to redds and egg-to-fry survival, while warmer summer temperatures may impact populations by moving more fish out of the main-stem and larger tributaries (where we surveyed) and/or through increased mortality (Hunt 1992; Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Mean monthly air temperatures in north-central Idaho were > 2 °C below normal during the winter of 2016 - 2017, and mean monthly summer air temperatures have been above normal every year except one since 1996 (NOAA 2021). These changes may have impacted fish populations in the Lochsa River through increased mortality and/or by moving fish out of our survey areas. Due to the importance of this fishery, this decline in density across all management sections warrants continued monitoring. Additional surveys will allow for a more thorough statistical analysis of population trends.

Mean density of all sizes of WCT, and WCT > 305 mm, was lower in the Mouth to WGB survey section than in other sections in 2013, and among the lowest in 2017. This is likely due to a combination of allowable harvest (the only main-stem Lochsa River section with harvest allowed) and warmer water temperatures in the most downstream river section. Densities have been highest in the uppermost survey sections, even though Colt Killed creek has allowable harvest. Although we do not have harvest estimates for WCT in the Lochsa River, harvest in this section is likely low, as most anglers are observed fishing the main-stem river. Thus, higher densities were likely a function of low harvest and cooler water temperature. These observations seem to suggest that during our traditional survey times WCT are utilizing colder upstream habitats more often. Because of this, it is important to add survey sites upstream of our traditional sites to determine if the decrease in mean density in our traditional sites reflects a true decline in population density or represents an upstream shift in habitat use.

Rainbow Trout

From 1975 to 2017, there has been a statistically significant declining trend in mean linear density of RBT in the main-stem Lochsa River. This trend in density has also been observed in the NFCR, Selway River, CDAR, and SJR (Hand et al. 2020; Ryan et al. 2020). We must caution that our data series for the main-stem Lochsa River contains a 30-year gap (1982-2012) and therefore may not properly represent trends in density. However, the ~97% decline observed in 2013 and 2017 compared to previous surveys is alarming. Declining trends in RBT have been primarily been attributed to hybridization, increasing temperatures from climate change, and habitat degradation (Zoellick 2005; Meyer et al. 2014; Muhlfeld et al. 2015). While these may be contributing factors in the Lochsa River, a major factor is likely the change in hatchery stocking practices during this time period. From 1968 to 1990, IDFG annually stocked ~11,000 RBT > 150 mm into the Lochsa River. No RBT have been stocked since, which would contribute to the decline in densities observed in surveys conducted since 1990. We have previously attributed declines in RBT within the North Fork Clearwater River drainage to the loss of steelhead from the construction of Dworshak dam (Pettit 1976; Hand et al. 2016). Declines in steelhead smolt out-migration, and adult returns have been observed throughout the Clearwater River basin, and specifically within the Lochsa River basin (Dobos et al. 2020; Feeken et al. 2020). Thus, recent declines in steelhead runs are also impacting RBT densities in the Lochsa River. Additional surveys will allow for a more thorough statistical analysis of trends in RBT density in the future.

Few RBT > 305 mm have been observed in the Lochsa River basin. This size distribution is similar to the Selway River where ~2% of RBT observed and caught by angling are > 305 mm (*see* Selway River chapter in this report). This is comparable to juvenile steelhead out-migrating from Fish Creek (major tributary of the Lochsa River) from 1995 to 2017, which had a maximum size of ~220 mm (Dobos et al. 2020). Additionally, up to 17% of hatchery steelhead smolts may residualize and remain in a river system as resident fish (McMichael et al. 1997; Hausch and Melnychuk 2012). Thus, the lack of larger RBT in the Lochsa River drainage is to be expected, as many of those observed in surveys are likely juvenile and residualized steelhead. With so few large fish, the population in the Lochsa River basin is mostly unavailable to the harvest fishery and would therefore be only minimally affected by angling.

Rainbow Trout density within survey sections is most likely influenced by water temperature, as harvest is not a primary contributing factor. Density has been highest in Colt Killed and Crooked Fork creeks, the upstream-most sections, which are cooler than downstream sections. The change in densities between these two survey sections from 2013 to 2017 was likely due to annual variation in distribution, as densities estimated during NPM surveys in these years were more stable (ranged from 0.8 to 1.1/100 m^2; Stiefel 2014; Putnam et al. 2018). The NPM surveys include more snorkel transects and cover larger areas of each drainage. Thus, they are more likely to account for annual variation in distribution.

Mountain Whitefish

Mean density of MWF (all sizes) declined from 2013 to 2017 in all survey sections except Colt Killed Creek. While our data in the Lochsa River is short-term, declines in MWF densities have been observed in the main-stem of other northern Idaho rivers, including the SFCR, Selway River, CDAR, and SJR (*See* Selway River and SFCR chapters in this report; Ryan et al. 2020). The only other large data set we have on MWF in the Lochsa River basin is from an intensive snorkel survey conducted in 2003, where the mean density (2.2/100 m^2) was higher than both 2013 and 2017 (Hand et al. 2008). While not directly comparable (due to not utilizing the same snorkel transects as other surveys), it does support the downward trend in average MWF density in the main-stem Lochsa River.

Long-term declines in MWF populations have been documented in other locations across the southern portion of their range as well, including the Big Lost River and Kootenai River, Idaho, the Yampa River, Colorado, and the Madison River, Montana (Paragamian 2002; IDFG 2007; Boyer 2016). Although these surveys occurred during a different time period, and thus are not directly comparable to our data, they suggest that habitat alteration, irrigation, nonnative fish interactions, disease, or harvest may be contributing to declines in MWF populations (IDFG 2007; Boyer 2016). While the direct cause of these declines has not been identified, these declines have been linked to occurrences of low flows and higher water temperatures (Brinkman et al. 2013). As discussed above, severe winter conditions and warmer summer temperatures are known to negatively impact salmonids. As such, these temperature changes would be expected to impact MWF. Increasing temperatures would also likely drive fish farther upstream and into deeper holes and tributaries as they seek cooler water, which could affect observations of MWF.

Smallmouth Bass

Smallmouth Bass were not observed in the Lochsa River during sampling in 2013 and 2017. In contrast, they have been observed in the lower reaches of other Clearwater River tributaries, including the North Fork Clearwater and South Fork Clearwater rivers (Hand et al. 2020; *see* South Fork Clearwater River chapter in this report). The lower North Fork Clearwater River is at a higher elevation (503 m) than the lower Lochsa River (448 m), suggesting that SMB could utilize the lower Lochsa River. Smallmouth Bass colonization of salmonid spawning and rearing habitat has been documented throughout the Columbia River Basin (Lawrence et al. 2014; Rubenson and Olden 2017). This includes locations at elevations higher than the Lochsa River, such as the Flathead River (790 m), Montana, the Salmon River (911 m) near Idaho Falls, Idaho, and the Owyhee River (1,030 m) upstream of Owyhee Reservoir (Rubenson and Olden 2017). Potential increases in SMB distribution is of concern, as these non-native fish can be a substantial predator of juvenile anadromous fish and could impact resident fish populations as well (Tabor et al. 1993; Naughton et al. 2004). At this time, few if any SMB appear to occur in the Lochsa River system. However, future surveys should continue to evaluate the distribution of these non-native fish, as they may experience a climate-mediated spread throughout the upper Clearwater River system (Rahel and Olden 2008).

MANAGEMENT RECOMMENDATIONS

1. Continue to evaluate trends in density and their size structure of game fishes in the Lochsa River drainage on a two year on, two year off basis, and assess whether season and limits, or environmental factors play a role in the trends that are being observed.

2. Add additional snorkel sites in upstream areas to evaluate whether populations are declining or fish are moving into these areas due to environmental conditions.

Table 8. Comparisons of Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) densities (fish/100 m) determined through snorkel surveys conducted in the Lochsa River, Idaho, from 1975 to 2017.

River section	Year	WCT	RBT
Mouth of Locsha River to Fish Creek	1975	0.00	0.37
	1976	0.16	0.04
	1977	0.08	4.23
	1978	0.11	2.56
	1979	0.13	4.38
	1980	0.33	0.66
	1981	0.50	11.00
	2013	0.84	0.08
	2017	0.64	< 0.01
Fish Creek to Lake Creek	1975	0.00	2.50
	1976	0.07	2.70
	1977	0.08	7.60
	1978	1.17	4.46
	1979	0.41	1.18
	1980	5.00	11.00
	1981	3.75	10.25
	2013	6.09	0.04
	2017	0.96	0.01
Lake Creek to Crooked Fork Creek	1975	0.00	1.14
	1976	0.00	3.72
	1977	1.15	0.00
	1978	2.20	0.00
	1979	3.04	0.68
	1980	8.00	0.00
	1981	6.67	0.00
	2013	10.14	< 0.01
	2017	3.32	0.01
Overall	1975	0.00	1.34
	1976	0.08	2.15
	1977	0.19	2.60
	1978	0.82	2.89
	1979	0.68	2.61
	1980	3.73	4.67
	1981	4.00	7.00
	2013	3.88	0.02
	2017	1.40	0.01

Table 9. Percent of Westslope Cutthroat Trout > 356 mm observed in snorkel surveys conducted in the main-stem Lochsa River, Idaho, in 2013 and 2017.

Survey section	2013	2017
Mouth - Wilderness Gateway Bridge	59	47
Wilderness Gateway Bridge - Lake Cr.	45	19
Lake Cr. - Crooked Fork Cr.	50	60
Crooked Fork Cr.	59	7
Colt Killed Cr.	52	40
Main-stem mean	49	47
Drainage mean	50	43

Table 10. Intrinsic rate of population change (r_{intr}) for Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) density (fish/100 m) in the Lochsa River basin, Idaho, from 1975 to 2017. Significance was set at α = 0.10.

	r_{intr}	90% CI	
Species	estimate	lower	upper
WCT	0.056	-0.016	0.128
RBT	-0.139	-0.171	-0.108

Table 11. Number and density of fish observed while snorkeling transects in the Lochsa River drainage, Idaho, during 2017.

Survey section	Transect name	Area (m^2)	Temp (°C)	Visability (m)	Number of fish WCT	RBT	MWF	Density (fish/100 m^2) WCT	RBT	MWF
Mouth to Wilderness Gateway Bridge	LR01	13,480	21	3.0	6	0	0	0.04	0.00	0.00
	LR02	21,019	17	3.3	7	0	0	0.03	0.00	0.00
	LR03	36,495	21	3.0	12	0	5	0.03	0.00	0.01
	LR04	15,744	19	3.1	0	0	9	0.00	0.00	0.06
	LR05	20,300	19	2.4	2	0	9	0.00	0.00	0.04
	LR06	25,211	20	2.8	7	0	4	0.03	0.00	0.02
	LR07	13,020	20	3.5	3	0	0	0.02	0.00	0.00
	LR08	10,872	19	2.7	5	6	11	0.05	0.06	0.10
	LR09	3,300	20	2.2	0	3	0	0.00	0.09	0.00
	LR10	12,462	18	2.8	1	0	3	0.01	0.00	0.02
	LR11	4,498	17	2.7	3	0	6	0.07	0.00	0.13
	LR12	6,534	---	3.0	3	1	5	0.05	0.02	0.08
	LR13	10,810	19	3.0	2	0	6	0.02	0.00	0.06
Wilderness Gateway Bridge to Lake Creek	LR14	7,118	---	1.7	3	8	2	0.04	0.11	0.03
	LR15	12,600	---	3.7	7	3	17	0.06	0.02	0.13
	LR16	5,980	---	2.8	0	0	11	0.00	0.00	0.18
	LR17	2,520	---	4.4	1	1	3	0.04	0.04	0.12
	LR18	5,343	---	---	7	8	5	0.13	0.15	0.09
	LR19	15,563	---	3.7	3	0	5	0.02	0.00	0.03
	LR20	13,570	---	5.0	7	1	18	0.05	0.01	0.13
	LR21	11,520	---	4.7	3	2	45	0.03	0.02	0.39
Lake Creek to Crooked Fork Creek	LR22	9,739	---	4.5	6	6	0	0.06	0.06	0.00
	LR23	8,100	19	2.4	18	0	20	0.22	0.00	0.25
	LR24	4,150	19	2.4	2	0	0	0.05	0.00	0.00
	LR25	5,334	16	2.8	6	11	9	0.11	0.21	0.17
	LR26	3,760	19	2.4	3	2	19	0.08	0.05	0.51
	LR27	4,340	19	2.4	27	9	40	0.62	0.21	0.92
	LR28	11,600	---	3.8	10	0	12	0.09	0.00	0.10
Crooked Fork Creek	CFC01	2,639	---	4.2	0	6	0	0.00	0.23	0.00
	CFC02	2,100	---	5.3	3	0	27	0.14	0.00	1.29
	CFC03	1,221	---	3.7	5	42	8	0.41	3.44	0.66
	CFC04	648	---	4.6	4	8	3	0.62	1.24	0.46
	CFC05	1,361	---	3.7	3	48	19	0.22	3.53	1.40
Colt Killed Creek	CKC01	3,300	---	3.8	3	0	0	0.09	0.00	0.00
	CKC02	3,984	---	4.4	0	1	27	0.00	0.03	0.68
	CKC03	2,588	---	3.0	0	2	1	0.00	0.08	0.04
	CKC04	4,836	---	2.6	1	0	9	0.02	0.00	0.19
	CKC05	6,200	---	3.3	6	5	14	0.10	0.08	0.23
	Mean				4.7	4.6	9.8	0.09	0.25	0.22
	90% CI				0.0	0.2	0.1	0.04	0.21	0.09

WCT - Westslope Cutthroat Trout; RBT - Rainbow Trout; MWF - Mountain Whitefish.

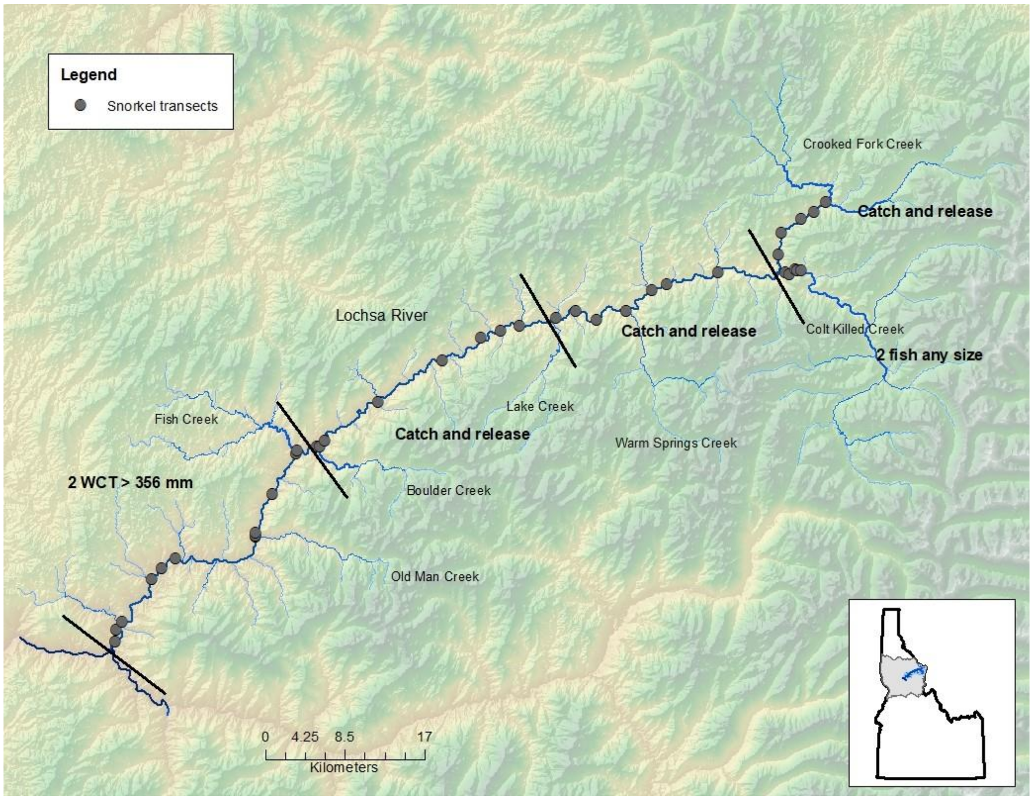

Figure 32. Map showing locations of snorkel transects surveyed in the Lochsa River drainage, Idaho, in 2017. Black bars delineate survey sections.

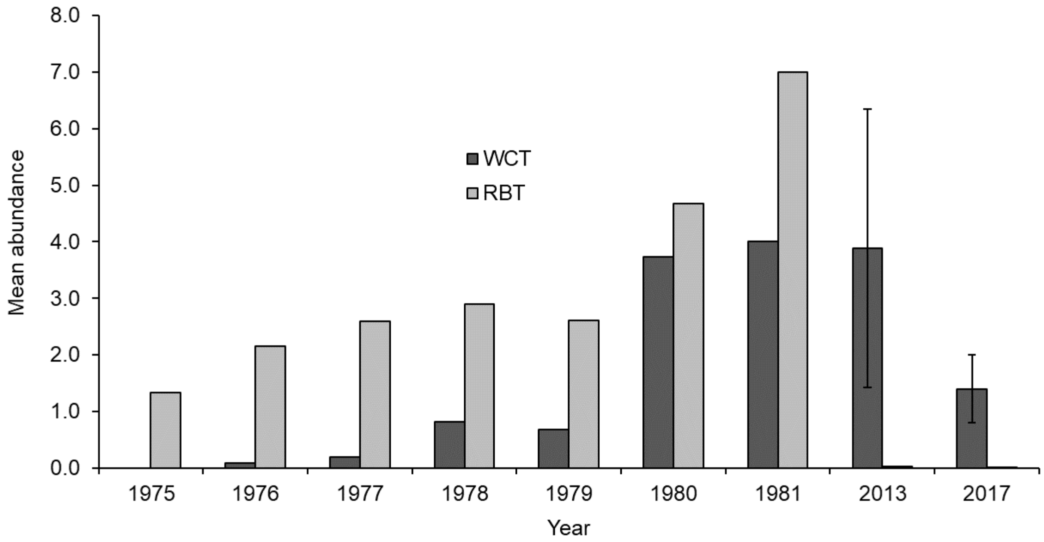

Figure 33. Comparisons of mean density (fish/100 m) of Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) observed during snorkel surveys of the main-stem Lochsa River, Idaho, from 1975 to 2017. Error bars represent 90% confidence intervals, and could only be calculated for 2013 and 2017.

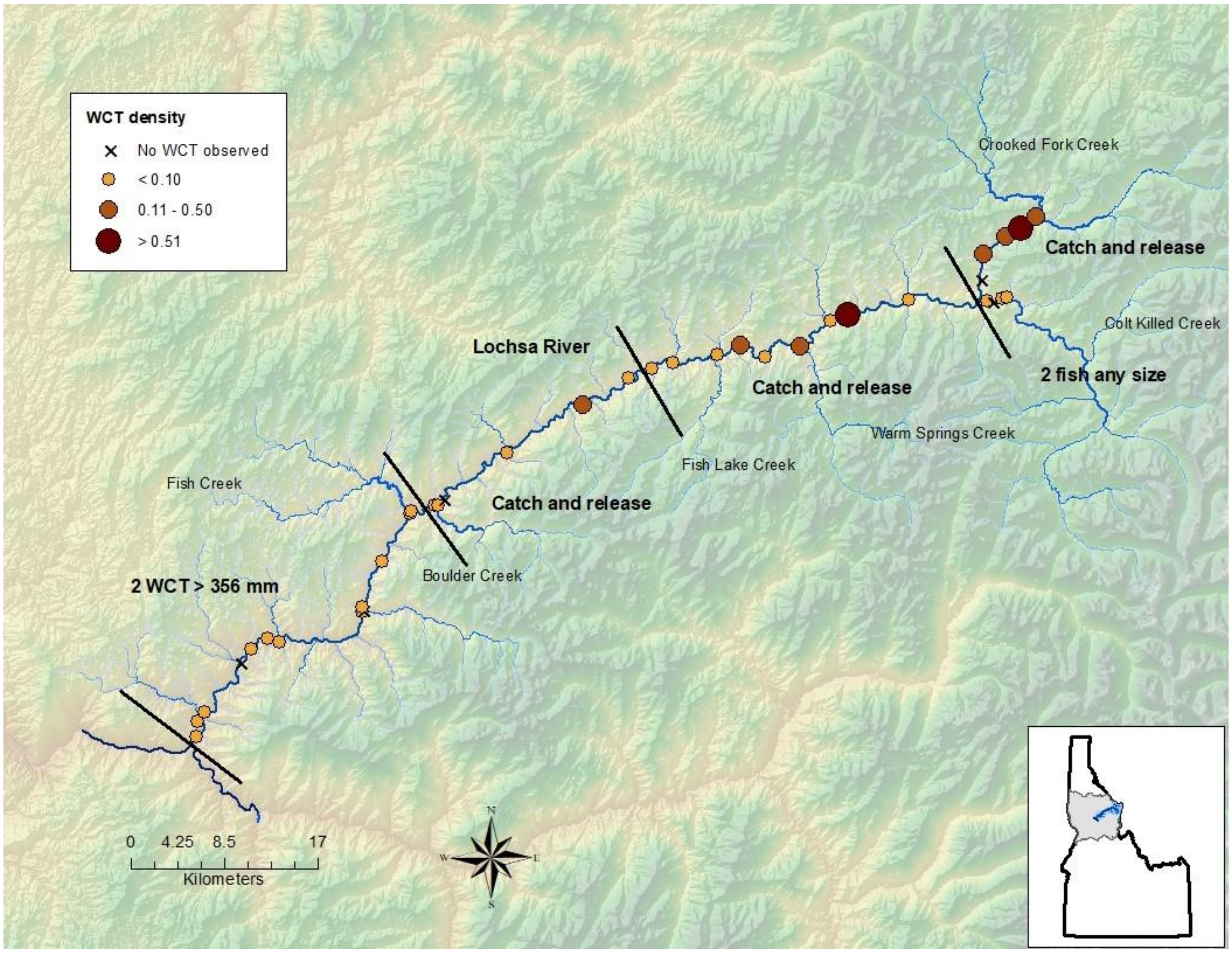

Figure 34. Densities (fish/100 m^2) of Westslope Cutthroat Trout (WCT) observed at each snorkel transect surveyed in the Lochsa River basin, Idaho, in 2017. Black bars delineate survey sections.

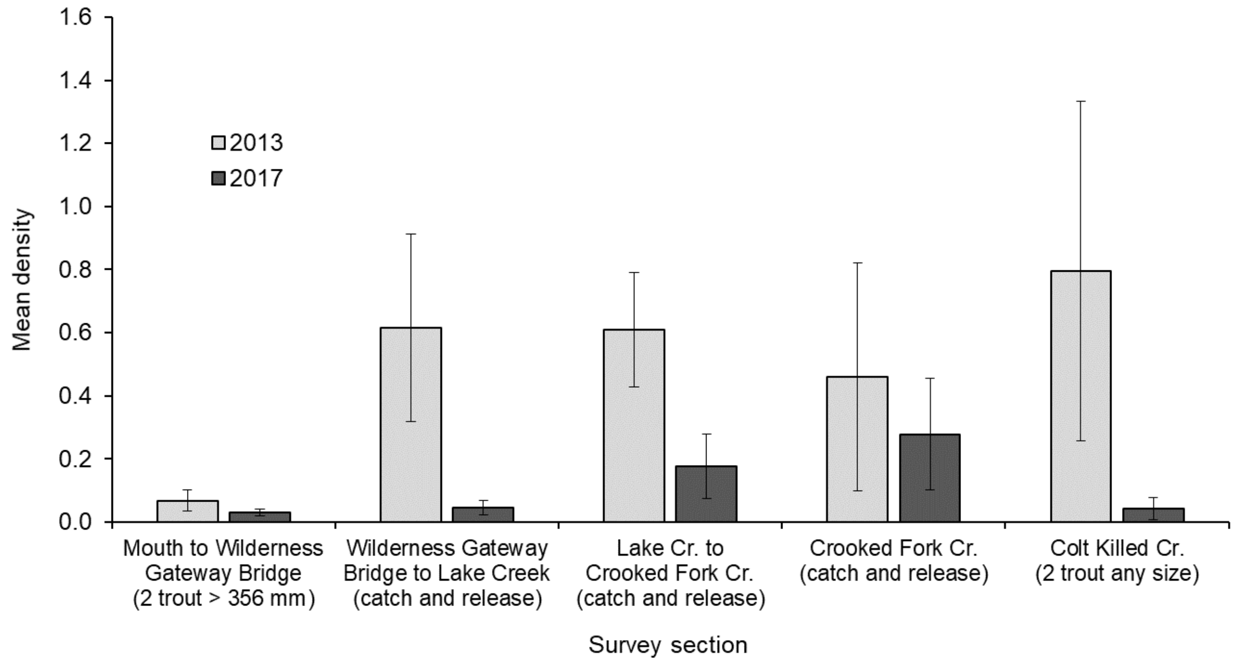

Figure 35. Comparisons of mean densities (fish/100 m^2) of Westslope Cutthroat Trout in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys in 2013 and 2017. Error bars represent 90% confidence intervals.

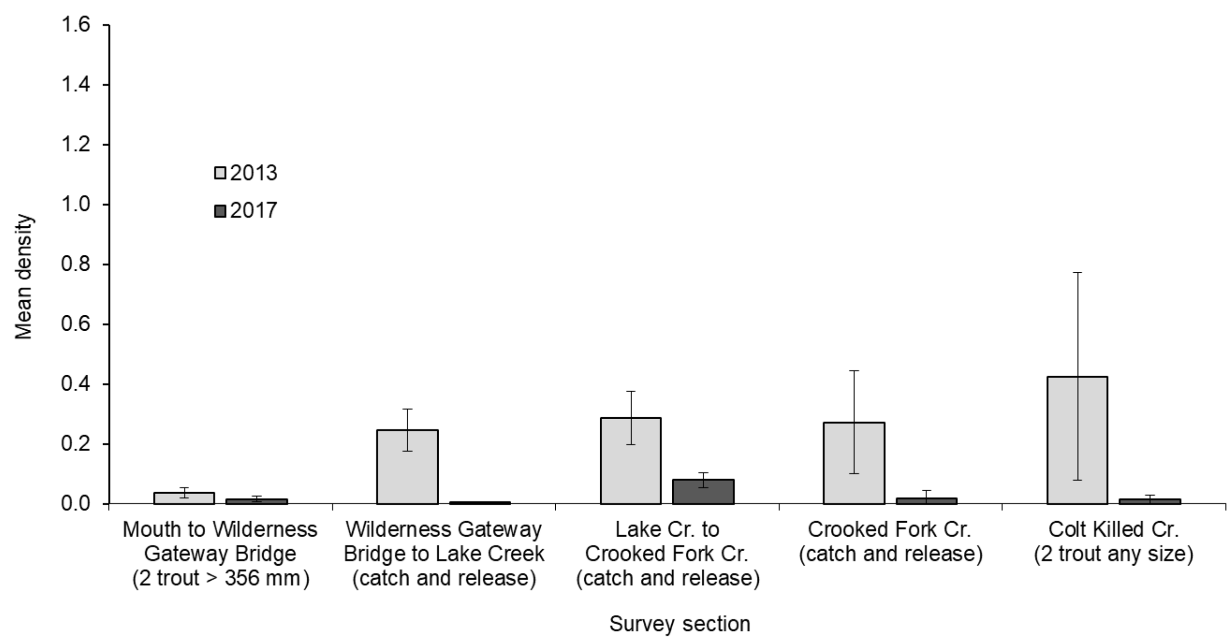

Figure 36. Comparisons of mean densities (fish/100 m^2) of Westslope Cutthroat Trout > 305 mm in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys in 2013 and 2017. Error bars represent 90% confidence intervals.

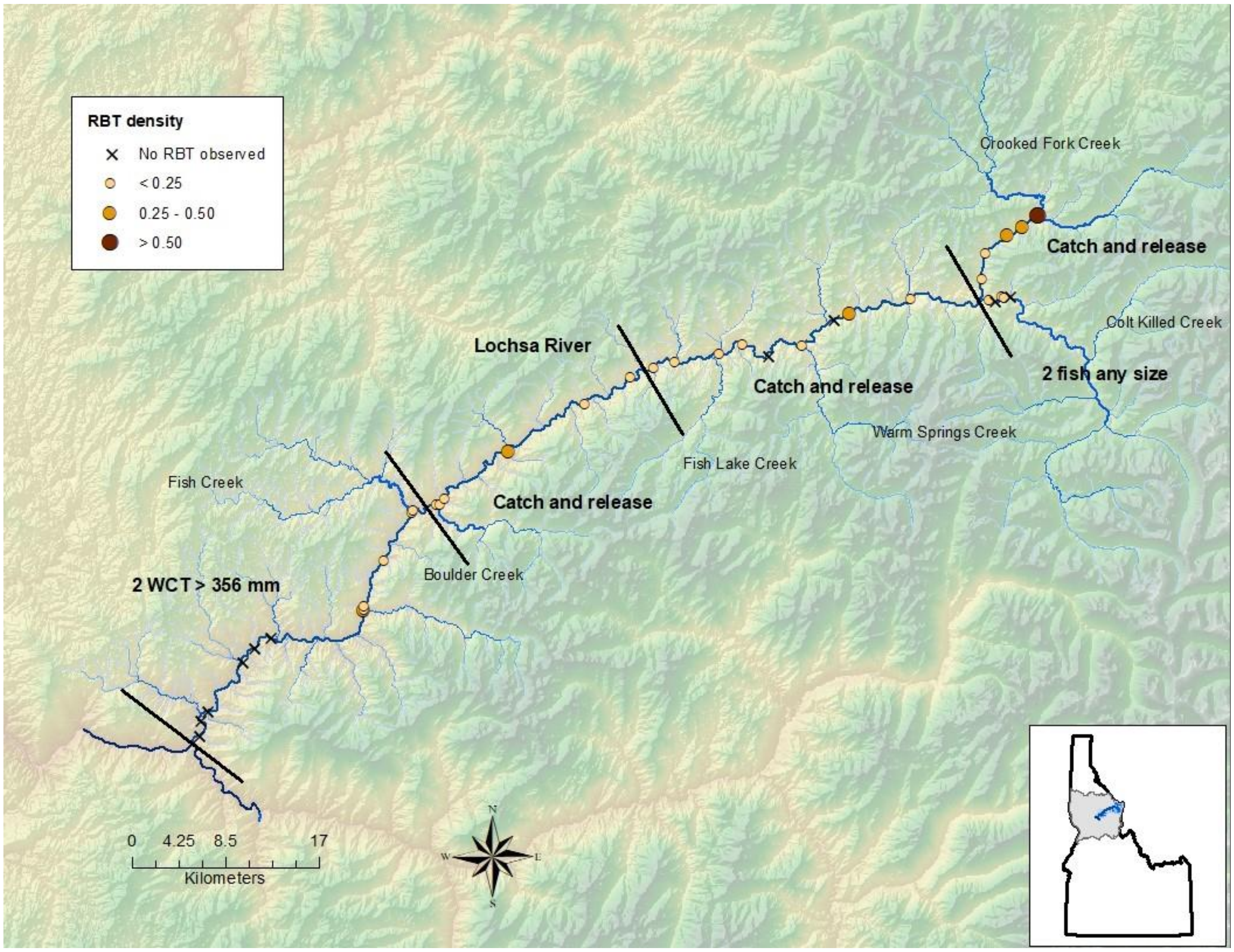

Figure 37. Densities (fish/100 m^2) of Rainbow Trout (RBT) observed at each snorkel transect surveyed in the Lochsa River basin, Idaho, in 2017.

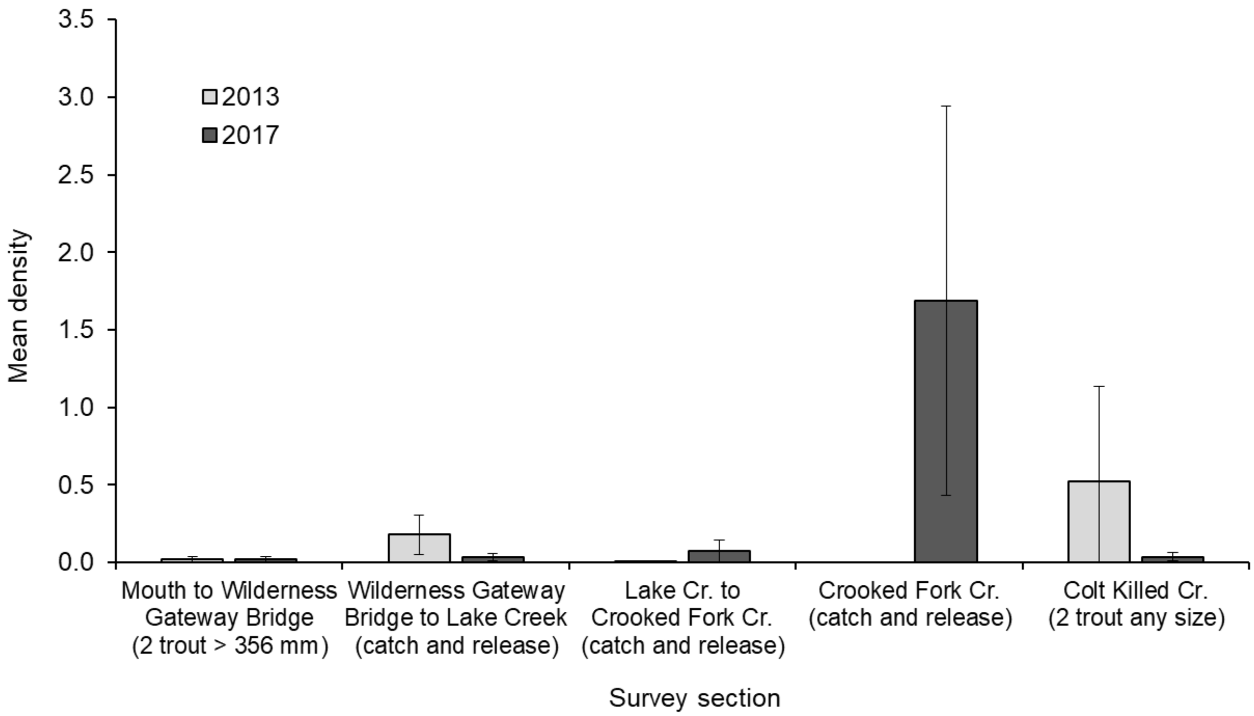

Figure 38. Comparisons of mean densities (fish/100 m^2) of Rainbow Trout in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys in 2013 and 2017. Error bars represent 90% confidence intervals.

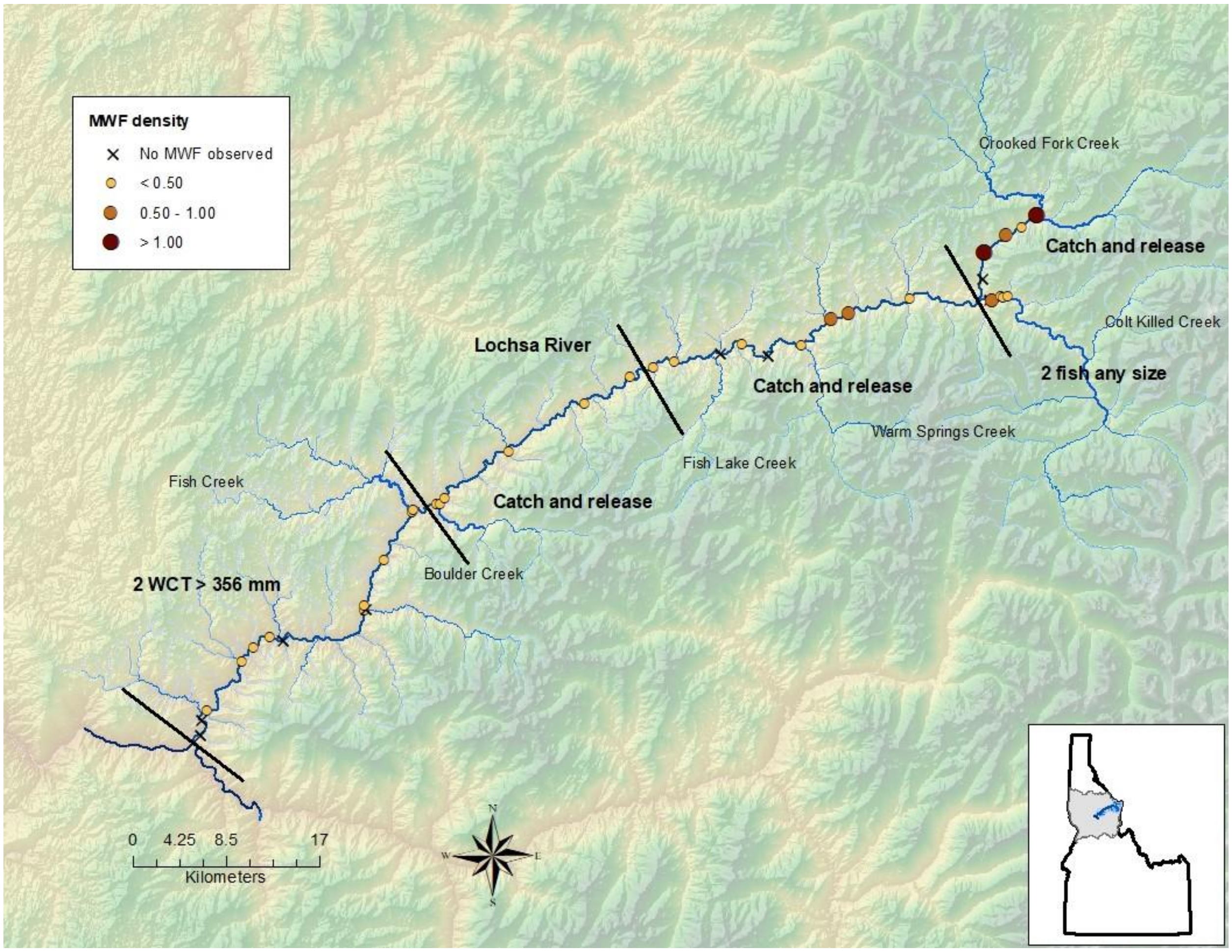

Figure 39. Densities (fish/100 m^2) of Mountain Whitefish (MWF) observed at each snorkel transect surveyed in the Lochsa River basin, Idaho, in 2017.

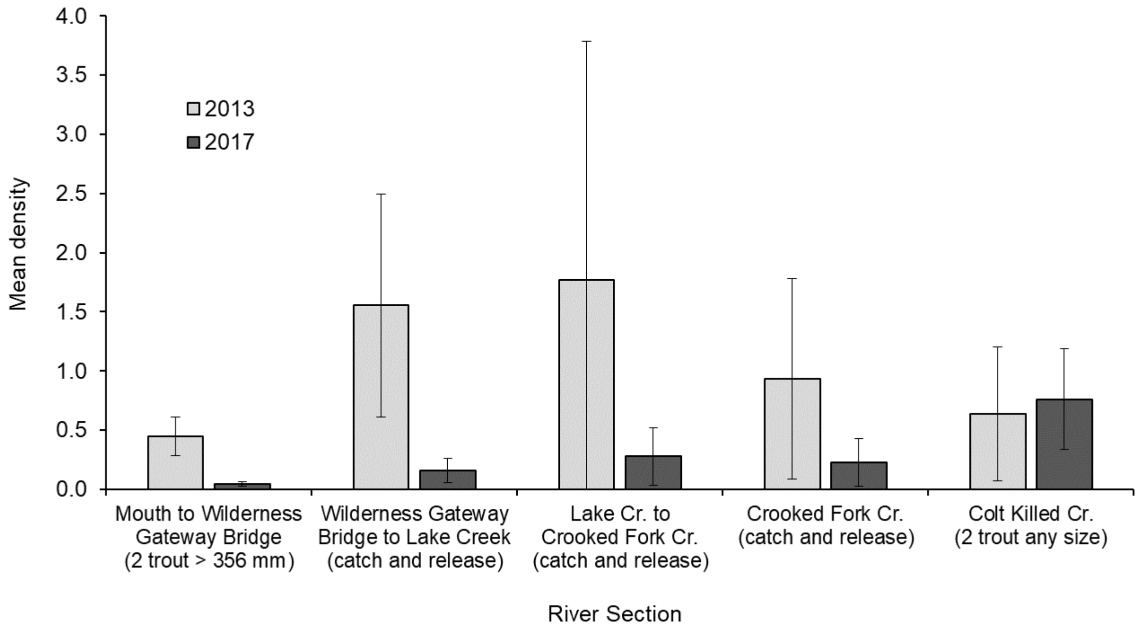

Figure 40. Comparisons of mean densities (fish/100 m^2) of Mountain Whitefish in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys in 2013 and 2017. Error bars represent 90% confidence intervals.

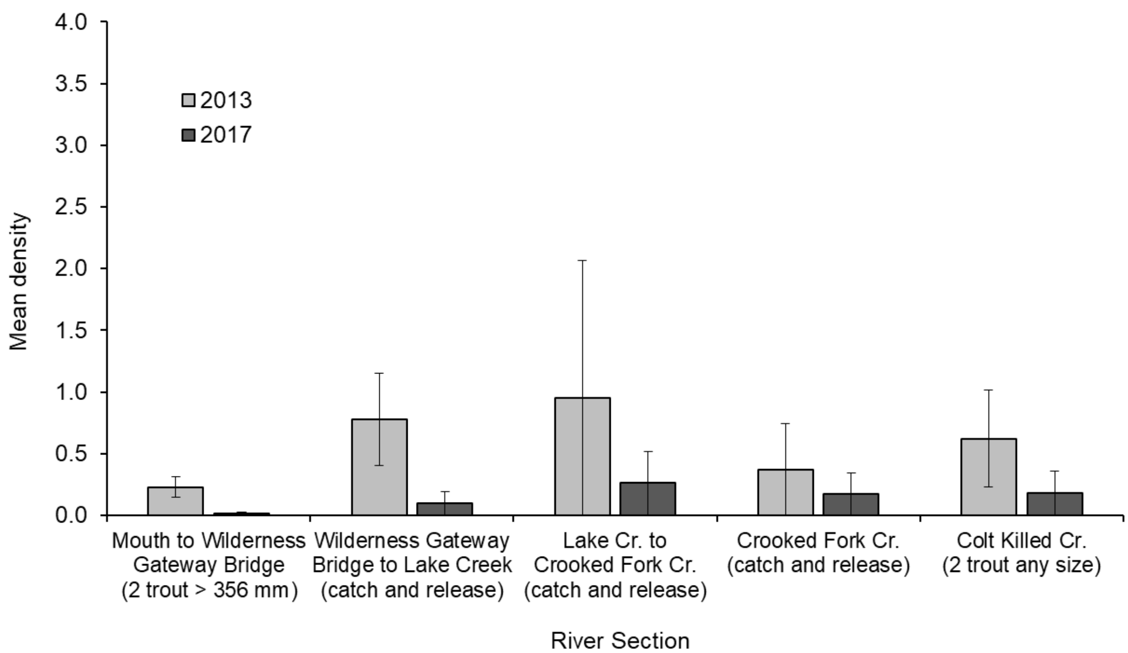

Figure 41. Comparisons of mean densities (fish/100 m^2) of Mountain Whitefish > 305 mm in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys in 2013 and 2017. Error bars represent 90% confidence intervals.

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SALMON RIVER FISHERY SURVEYS

ABSTRACT

The main-stem Salmon River and tributaries are seldom surveyed due to the difficult access of this remote area. To increase our understanding of the fisheries in this area we conducted snorkel surveys of tributaries, and hook-and-line surveys of the main-stem Salmon River between Corn and Allison creeks to evaluate density and distribution of fishes in this river reach. We also conducted backpack electrofishing on the main-stem to evaluate density and distribution of Pacific Lamprey *Entosphenus tridentatus* (PL). Snorkel surveys indicated that Rainbow Trout *Oncorhynchus mykiss* and Westslope Cutthroat Trout *O. clarkii lewisi* were distributed throughout these tributaries, with mean densities higher than observed in other Salmon River tributaries surveyed in 2017. Mean densities of Mountain Whitefish *Prosopium williamsoni* were lower than for other Salmon River tributaries surveyed in 2017. The hook-and-line catch rate for all species combined was lower than the mean catch rate for surveys of the Selway and Middle Fork Salmon rivers from 2012 - 2017. Pacific Lamprey were present at 92% of survey sites, however density appeared lower than densities in previous surveys of the Salmon River and Clearwater River drainages conducted from 2003 to 2018. With < 6% of Westslope Cutthroat Trout, and no Rainbow Trout or Mountain Whitefish > 300 mm observed in snorkel surveys, quality-size fish appear to be rare. Northern Pikeminnow *Ptychocheilus oregonensis* was the predominant species caught by hook-and-line in our survey, while Westslope Cutthroat Trout was the predominant species caught in other wilderness river surveys. The difference in catch rate and species composition was primarily due to higher summer water temperatures of the main-stem Salmon River. The lower densities of PL were likely a combination of the lower densities that occur farther downstream in the drainage, and our skill level related to electrofishing technique and determining preferred PL habitat compared to those conducting previous surveys. We should improve our sampling technique and ability to delineate PL habitat, and estimate density as fish/100 m^2 in future surveys to allow for direct comparison with historic data. We recommend conducting all of these surveys of this stretch of the Salmon River every few years to track trends in fish communities over time.

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INTRODUCTION

The Salmon River drainage is a large and remote, and supports a diversity of wild salmonids including steelhead and Chinook Salmon in the main-stem and Rainbow Trout *Oncorhynchus mykiss* (RBT) and Westslope Cutthroat Trout *O. clarkii lewisi* (WCT) in tributaries. The tributary streams in the Salmon River canyon represent the largest production area for wild steelhead in the Salmon River basin, and fishing opportunities in these tributaries are supported primarily by juvenile steelhead (IDFG 2019). Our understanding of fish populations in the much of the drainage is limited due to sporadic and incomplete sampling. To increase our understanding of the fish populations in this area, we conducted snorkel surveys in tributaries, and hook-and-line surveys in the main-stem Salmon River between Corn and Allison creeks. Establishing survey transects in some of these tributaries that have been rarely sampled will provide baseline data for future surveys.

Pacific Lamprey *Entosphenus tridentatus* (PL), a native anadromous species of the north Pacific, were once abundant throughout their range but have experienced drastic declines in recent decades (Beamish and Northcote 1989; Kostow 2002; Moser and Close 2003). In the Salmon River, PL have experienced significant declines since the 1960s but are still present based on sporadic surveys (USACE 2009; Hyatt et al. 2007). Because PL are not recognized as a game fish species, little attention has been given to their status. To improve our understanding of PL density and distribution, we conducted electrofishing surveys on the main-stem Salmon River between Corn and Allison creeks.

OBJECTIVES

1. Determine which fishes are present and their relative densities in tributaries of the Salmon River between Corn and Allison creeks.

2. Assess the relative density of fishes that are susceptible to hook-and-line sampingin the main-stem Salmon River between Corn and Allison creeks during the summer.

3. Assess the density and distribution of Pacific Lamprey in the main-stem Salmon River between Corn and Allison creeks.

STUDY AREA

The Salmon River is classified as a Wild and Scenic River, flowing 660 km from its headwaters in south central Idaho to its confluence with the Snake River in west central Idaho. The drainage covers an area of over 1.6 million ha. This survey was conducted on the main-stem Salmon River and tributaries between Corn and Allison creeks which is located in the central section of the main-stem river and consists of ~119 km of roadless river (Figure 43). This section of the Salmon River is commonly referred to as the Salmon River Canyon, and is classified "wild" under the Wild and Scenic Rivers System. It supports a wide range of recreational opportunities including rafting, jet boating, and angling.

METHODS

Field sampling

Snorkel survey

Snorkel surveys were conducted from August 9 to 19, 2017 in tributaries of the Salmon River between Corn and Allison creeks. A total of 15 transects were surveyed on ten tributaries (Figure 43). Transects were selected based on a generalized random-tessellation stratification design (Stevens and Olsen 2004). A more detailed discussion of this selection process can be found in Stiefel et al. (2015). We selected transects that were within one mile of the main-stem Salmon River to ensure they could be surveyed in a timely manner during a float trip of the main-stem Salmon River. Surveys were conducted by one or two snorkelers, depending on the width of each transect, using the standard snorkeling methodologies outlined in Apperson et al. 2015. A single snorkeler was used only when the entire wetted width of the stream could be effectively observed by one person. Surveys were conducted moving upstream. All fish observed were counted, and length was estimated to the nearest inch for all game species. Non-game species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson 50-m, reel-style, fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. The distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead and resident Rainbow Trout are indistinguishable, and are collectively referred to as “RBT” in this chapter.

Hook-and-line survey

Hook-and-line sampling was conducted from August 14 to 19, 2017, on the main-stem Salmon River between Corn and Allison creeks. Each person fishing kept track of where, when, and how long they fished, what species they caught, how long it was (total length mm), and gear type used (lure, fly).

Pacific Lamprey survey

Backpack electrofishing was conducted from August 14 to 19, 2017, on 13 transects in main-stem Salmon River between Corn Creek and Vinegar Creek. Sampling was conducted with an ETS ABP-2 backpack electrofisher using recommended lamprey-specific settings (U.S. Fish and Wildlife Service 2010). Potential sample transects were evaluated based upon physical habitat characteristics, and sampled as time allowed (Claire 2003). Transects were marked in a handheld GPS unit with a unique name, and latitude and longitude were recorded in decimal degrees WGS84 format. Transect length (m), predominant substrate, and habitat type were recorded. Pacific Lamprey sampled were measured for total length (mm), and additional PL observed but not captured were enumerated.

Data analysis

Snorkel survey

To evaluate density and distribution of fishes, we estimated mean areal density (fish/100 m^2) for each species observed in each transect. No statistical comparisons were made to previous surveys of these tributaries due to low sample size and/or inconsistencies in transect locations. We compared areal densities to other Salmon River and Selway River tributaries surveyed in 2017 (*see* Selway River section of this report; Putnam et al. 2018). Locations of where fishes (WCT, RBT, and MWF) were observed and their relative areal density were visually displayed by plotting density representative circles on maps of the survey area using GIS software.

Hook-and-line survey

We assessed the relative density of fishes that were susceptible to hook-and-line fishing by calculating catch rates (fish/h) for all species caught.

Pacific Lamprey survey

Pacific Lamprey linear density was evaluated by estimating the number/100 m of transect length. Linear density estimates calculated for 2017 were not compared statistically to previous surveys, as those surveys used areal densities (#/100 m^2), and survey transect locations in 2017 were not consistent with previous surveys. Locations of where PL were observed and their relative linear density were visually represented by plotting density representative circles on maps of the survey area using GIS software. Mean linear density was compared to other rivers within the Clearwater River and Salmon River drainages the Selway River (Hyatt et al. 2017; *see* Selway River section of this report).

RESULTS

Snorkel survey

Snorkel surveys resulted in the observation of RBT, Westslope Cutthroat Trout *O. clarkii lewisii* (WCT), Bull Trout *Salvelinus confluentus* (BT), Mountain Whitefish *Prosopium williamsoni* (MWF), Chinook Salmon *O. tshawytscha*, and Brook Trout *S. fontinalis* (BKT). Rainbow Trout were the most common species observed (7.75/100 m^2), with areal densities over eight times higher than any other species (Table 12). Westslope Cutthroat Trout were the second most common species (0.98/100 m^2; Table 12). Bull Trout and Brook Trout were the least common, with areal densities of both species < 0.01/100 m^2 (Table 12). Mean areal density of RBT was twice as high as other Salmon River tributaries, and over four times higher than Selway River tributaries. Mean areal density of WCT was similar, while MWF density was lower, than other Salmon River and Selway River tributaries (Table 13).

Eight-nine percent of the WCT, RBT, and MWF observed were < 200 mm in length (Table 14). Only 6% of WCT, and no RBT or MWF observed were > 300 mm.

Westslope Cutthroat Trout and RBT were widely distributed through the survey area, while MWF were observed only in tributaries from California Creek upstream. Distributions for each species are displayed in Figures 43 - 47.

Hook-and-line survey

The hook-and-line survey resulted in the capture of 115 Northern Pikeminnow *Ptychocheilus oregonensis*, 8 RBT, 5 WCT, 2 Smallmouth Bass *Micropterus dolomieu* (SMB), and 2 BT during an estimated 84.2 hours of angling effort (Table 15). The mean CPUE for all fish caught was 1.5 fish/h (Table 15).

Pacific Lamprey survey

The electrofishing survey resulted in a mean PL linear density of 19.7/100 m (Table 16). Mean density in 2017 appeared lower than the mean densities of 6 - 568 PL/100 m^2 in the Salmon River drainage from 2003 to 2006, and 137 - 1,536/100 m^2 in the Clearwater River drainage from 2001 to 2018 (Table 17). Mean PL length was 66 mm, with 55% < 60 mm (Figure 48). Pacific Lamprey were distributed throughout the survey area, and were present in 12 of the 13 transects (Figure 49). Water temperatures at these survey transects ranged from 17 to 20 °C, with a mean of 19 °C.

DISCUSSION

Snorkel survey

Westslope Cutthroat Trout were observed in all tributaries surveyed in 2017 except Allison Creek. This may be due to it being the smallest tributary we surveyed (~3 m width). The previous survey conducted in these tributaries in 2008 found WCT present in all of them including Allison Creek. The mean density of WCT observed in these surveys was higher than the mean density of other (both up and down stream) Salmon River tributaries snorkeled in 2017, but was lower than the 20-year mean for Selway River tributaries (Putnam et al. 2018; *see* Selway River chapter in this report). Although most of these tributaries are large enough for angling (> 5 m width), with only 6% of the WCT observed > 300 mm in length, these tributaries would provide little in the way of a resident fishery.

Rainbow Trout were observed in all tributaries surveyed in 2017 and were observed in all of these tributaries, except Allison Creek, during the previous survey in 2008. Jersey Creek had the highest density of RBT, which was four times higher than any other tributary. Mean density of RBT was higher than other (both up and down stream) Salmon River tributaries snorkeled in 2017 and the 20-year mean for Selway River tributaries (Putnam et al. 2018; *see* Selway River chapter in this report). Although most of these tributaries are large enough for angling (> 5 m width), with no RBT observed > 300 mm in length, these tributaries would provide little in the way of a resident fishery.

Mountain Whitefish presence appears to be influenced by tributary size, as they were observed in four the largest tributaries in 2017. They have been observed in all of these tributaries except Allison Creek, during at least one previous survey occurring from 1985 to 2008. They have not been observed in Allison Creek, the smallest tributary we surveyed, during any survey. Mean density of MWF observed in this survey was lower than other (both up and down stream) Salmon River tributaries snorkeled in 2017, but was similar to the 20-year mean for Selway River tributaries (Putnam et al. 2018; *see* Selway River chapter in this report).

Juvenile Chinook Salmon were observed in four of the tributaries surveyed in 2017. They have been observed in all of the tributaries surveyed (except Allison Creek), during at least one previous survey conducted from 1985 to 2008. Mean density of Chinook Salmon was lower than other Salmon River tributaries (both up and down stream) snorkeled in 2017, but similar to the 20-year mean for Selway River tributaries (Putnam et al. 2018; *see* Selway River chapter in this report).

Bull Trout were observed in Horse and Big Mallard creeks. They were not observed in these tributaries previously, but have been observed previously in Crooked (2001), Chamberlain (1994), and Jersey (2002) creeks. Mean density of BT in our survey was lower than other (both up and down stream) Salmon River tributaries snorkeled in 2017, but similar to the 20-year mean for Selway River tributaries (Putnam et al. 2018; *see* Selway River chapter in this report). Based on the range of water temperature and stream width of the tributaries where BT were observed, these characteristics do not appear to be the primary factors determining distribution of BT in these tributaries. This suggests they may be distributed in many of these tributaries, but at low densities.

Hook-and-line survey

The hook-and-line survey resulted in the capture of 132 fish, 87% of which were Northern Pikeminnow *Ptychocheilus oregonensis*. The catch rate (1.5 fish/h) for all species was lower than hook-and-line sampling conducted during float trips on the Selway River (3.0 - 4.3 fish/h) and Middle Fork Salmon River (2.8 - 5.8 fish/h) from 2012 to 2017 (*see* Selway River chapter in this report; Messner and Schoby 2019). This lower catch rate may be a function of water temperatures, as this section of the Salmon River averaged ~3 °C warmer than the mean water temperature for main-stem Selway River surveys conducted from 2012 to 2017. Additionally, all trout were caught at the mouth of tributaries, which generally have lower water temperatures. This indicates that warmer temperatures in the main-stem were a primary factor affecting catch rates (especially for trout). Few SMB were caught during this survey, despite the warmer water temperatures. Previous surveys in the Salmon River downstream of Hammer Creek resulted in the capture of numerous SMB.

Pacific Lamprey survey

Pacific Lamprey were present throughout the survey area, occurring in 92% of transects. While our data was not directly comparable to the areal density estimates from previous surveys, mean linear density was lower than those observed in previous surveys in Clearwater River tributaries, and all other sections of the Salmon River except the section downstream of the Little Salmon River (Hyatt et al. 2007; IDFG *unpublished data*). Density of PL in the Salmon River drainage appears to increase the farther upstream the survey section is located, and may be related to habitat quality/quantity. With our survey area occurring farther downstream in the drainage, we would therefore expect density of PL to be lower. However, another factor potentially influencing the lower density observed in our survey was our lower level of experience in sampling these fish compared to those conducting the previous surveys. Improving our technique and understanding of preferred PL habitat will ensure our data is more comparable to previous surveys. Future surveys should employ the same techniques used in historic surveys, and ensure similar habitat is surveyed, to allow for direct comparisons.

MANAGEMENT RECOMMENDATIONS

1. Continue to conduct surveys of the Salmon River and its tributaries every few years to track trends in fish communities over time.

Table 12. Fish density (fish/100 m^2) by transect, determined by snorkel surveys in tributaries of the Salmon River, Idaho, between Corn Creek and Allison Creek, during 2017.

						Density								
Stream	Transect name	Survey date	Transect length (m)	Transect area (m^2)	Temp (°C)	Westslope Cutthroat Trout	Bull Trout	Brook Trout	Mountain Whitefish	RBT	Trout fry	Chinook Salmon	Longitude	Latitude
Allison Creek	AC01	8/10/2017	73	224	15	0.00	0.00	0.00	0.00	5.36	0.00	0.00	-116.1641	45.4188
Allison Creek	AC02	8/9/2017	101	309	15	0.00	0.00	0.00	0.00	11.00	0.00	0.00	-116.1689	45.4527
Big Mallard	1	8/16/2017	65	551	15	0.18	0.18	0.00	2.36	7.80	0.00	5.99	-115.2736	45.5399
California Creek	2	8/18/2017	38	257	15	3.90	0.00	0.00	0.00	5.85	0.39	2.34	-115.7607	45.4482
Chamberlain Creek	MOUTH(L1)	8/15/2017	55	756	14	1.59	0.00	0.00	0.93	1.32	0.00	0.00	-114.9341	45.4541
Chamberlain Creek	RUN(2)	8/15/2017	45	610	12	0.00	0.00	0.00	1.15	0.16	0.00	0.00	-114.9372	45.4534
Crooked Creek	1	8/18/2017	35	613	15	0.00	0.00	0.00	1.63	1.63	0.00	0.00	-115.6625	45.4418
Crooked Creek	2	8/18/2017	63	714	16	0.14	0.00	0.00	0.14	7.00	0.00	0.56	-115.6552	45.4468
French Creek	FC01	8/10/2017	92	948	17	0.21	0.00	0.00	0.00	4.12	0.42	0.00	-116.0274	45.4231
French Creek	FC02	8/10/2017	100	1108	18	0.09	0.00	0.09	0.00	7.76	0.45	0.00	-116.0330	45.4022
Horse Creek	L1	8/14/2017	53	752	14	1.73	0.13	0.00	1.20	9.58	0.00	0.27	-114.7329	45.3976
Horse Creek	L2	8/14/2017	53	553	14	1.81	0.00	0.00	1.27	5.43	0.00	1.63	-114.7361	45.4010
Jersey Creek	1	8/17/2017	31	104	17	0.96	0.00	0.00	0.00	44.99	20.10	0.00	-115.4635	45.4184
Little Fivemile	2	8/17/2017	40	149	---	3.36	0.00	0.00	0.00	0.00	0.00	0.00	-115.4612	45.4165
Richardson	1	8/16/2017	25	140	13	0.71	0.00	0.00	0.00	4.29	0.00	0.00	-115.2623	45.5367
Mean						0.98	<0.01	<0.01	0.58	7.75	1.42	0.72		
90% CI						0.54	<0.01	<0.01	0.33	4.59	2.20	0.69		

Table 13. Mean density (fish/100 m^2) for Rainbow Trout (RBT), Westslope Cutthroat Trout (WCT), and Mountain Whitefish (MWF) determined by snorkel surveys in tributaries of the Salmon River and Selway River, Idaho, during 2017.

River	Density		
	RBT	WCT	MWF
Salmon River			
Corn Cr. - Allison Cr.	7.75	0.98	0.58
Other tributaries	3.94	0.71	1.61
Selway River	1.57	1.05	0.78

Table 14. Number of Rainbow Trout (RBT), Westslope Cutthroat Trout (WCT), and Mountain Whitefish (MWF) by length groups, as determined by snorkel surveys in tributaries of the Salmon River, Idaho, between Corn Creek and Allison Creek, during 2017.

Species	Length (mm)		
	< 200	200 - 300	> 300
RBT	397	43	0
WCT	44	7	3
MWF	48	6	0

Table 15. Catch-per-unit-effort (CPUE; fish/h) and mean length (mm) of fish caught by hook-and-line sampling on the Salmon River, Idaho, between Corn Creek and Alison Creek, from August 14 to 19, 2017.

Species	Number caught	CPUE	Mean length
Northern Pikeminnow	115	1.4	265
Rainbow Trout	8	0.1	197
Westslope Cutthroat Trout	5	0.1	322
Bull Trout	2	<0.1	339
Smallmouth Bass	2	<0.1	279
Total	132	1.5	

Table 16. Number of Pacific Lamprey captured and observed by transect, and linear density (fish/100 m), determined by backpack electrofishing surveys on the main-stem Salmon River, Idaho, between Corn and Allison creeks, during 2017.

Transect name	Survey date	Transect length (m)	Temp (°C)	Number of PL Captured	Number of PL Observed	Number of PL Total	Linear density	Longitude	Latitude
LAM001	8/14/2017	100	17	3	0	3	3	-114.7362	45.3947
LAM002	8/14/2017	75	18	2	0	2	3	-114.8068	45.3943
LAM003	8/14/2017	100	18	0	1	1	1	-114.8745	45.4195
LAM004	8/15/2017	10	19	5	6	11	110	-115.0321	45.5071
LAM005	8/15/2017	10	20	7	0	7	70	-115.1354	45.5406
LAM006	8/16/2017	20	18	3	0	3	15	-115.2582	45.5387
LAM007	8/17/2017	40	18	8	0	8	20	-115.3938	45.4727
LAM008	8/17/2017	30	19	0	2	2	7	-115.4401	45.4442
LAM009	8/17/2017	25	19	2	1	3	12	-115.4654	45.4161
LAM010	8/18/2017	25	19	0	1	1	4	-115.6062	45.3960
LAM011	8/18/2017	35	20	1	1	2	6	-115.6597	45.4296
LAM012	8/18/2017	35	20	0	1	1	3	-115.7426	45.4450
LAM013	8/19/2017	75	18	0	2	2	3	-115.8136	45.4663
Mean		45	19	2.4	1.2	3.5	19.7		
90% CI				1.3	0.7	1.4	14.9		

Table 17. Mean linear density (#/100 m) and areal density (#/100 m^2) of Pacific Lamprey sampled by electrofishing in the Clearwater River and Salmon River drainages, Idaho, from 2001 to 2018.

Date range	River	Reach	Linear density	Areal density
2005-2006	Salmon	MF Salmon		301
2003-2006	Salmon	Above SF Salmon		568
2004-2006	Salmon	Little Salmon to SF Salmon		192
2017	Salmon	Corn Cr. to Vinegar Cr.	20	
2003-2006	Salmon	Below Little Salmon		6
2018	Selway		540	
2002-2006	Lochsa			1,536
2001-2006	SF Clearwater			407
2001-2006	Red River			137

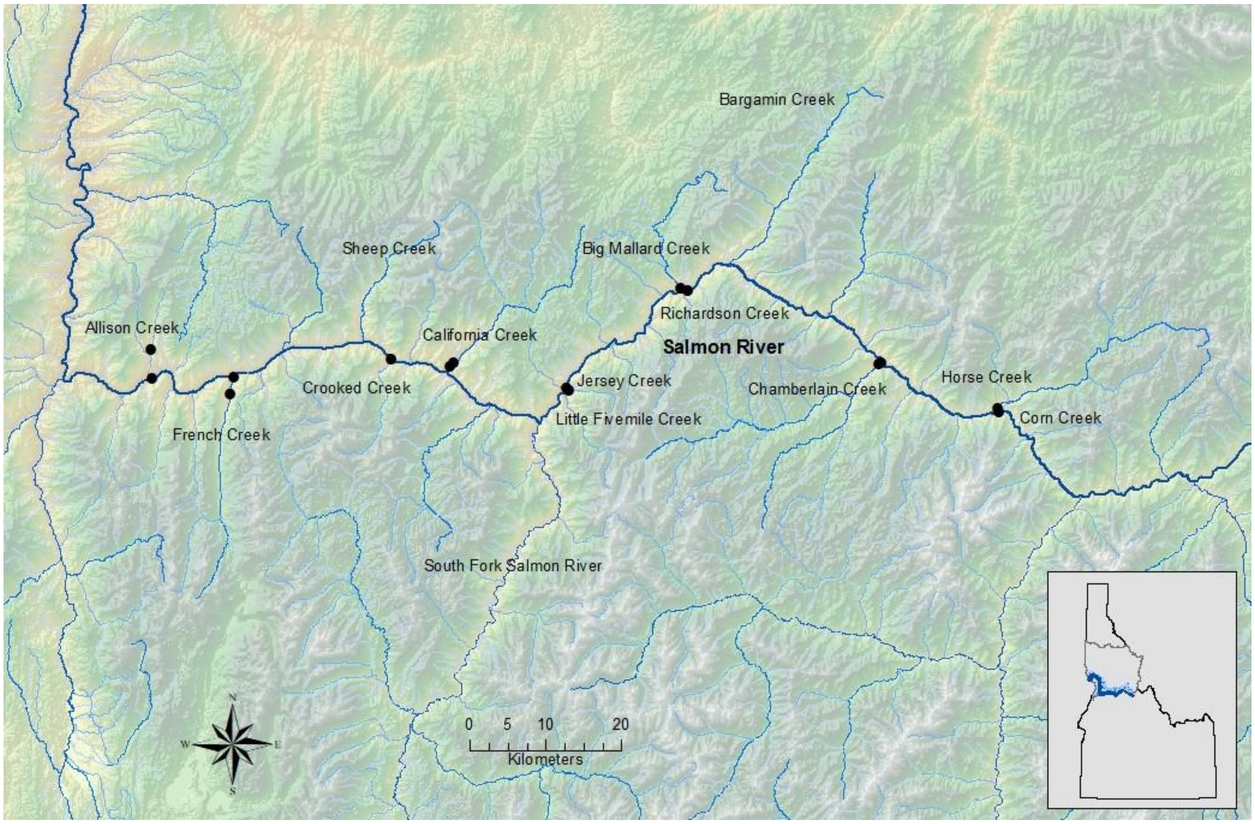

Figure 42. Map showing locations of snorkel transects surveyed in the Salmon River drainage, Idaho, in 2017.

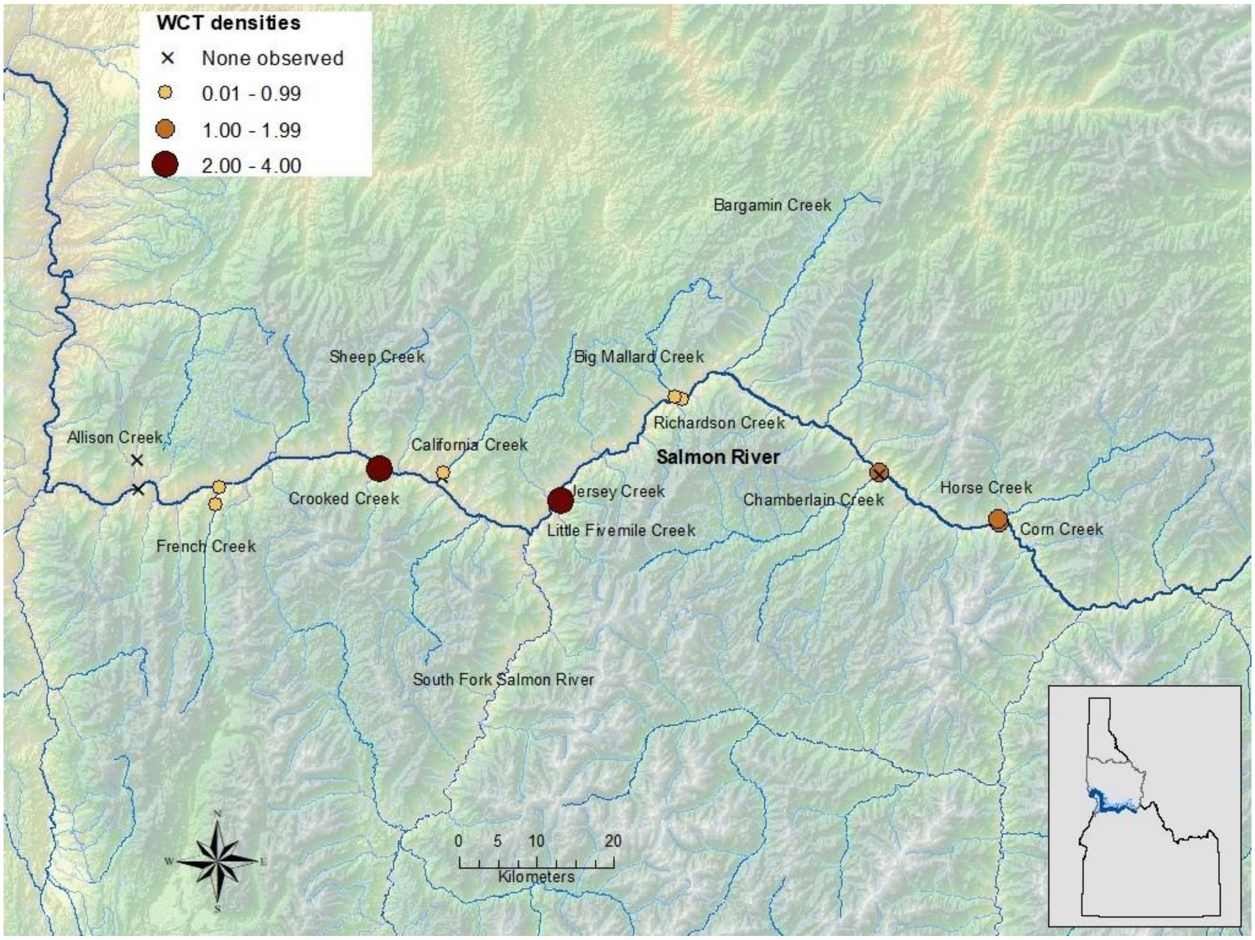

Figure 43. Areal density (fish/100 m^2) of Westslope Cutthroat Trout (WCT) observed in each snorkel transect surveyed in tributaries of the Salmon River, Idaho, during 2017.

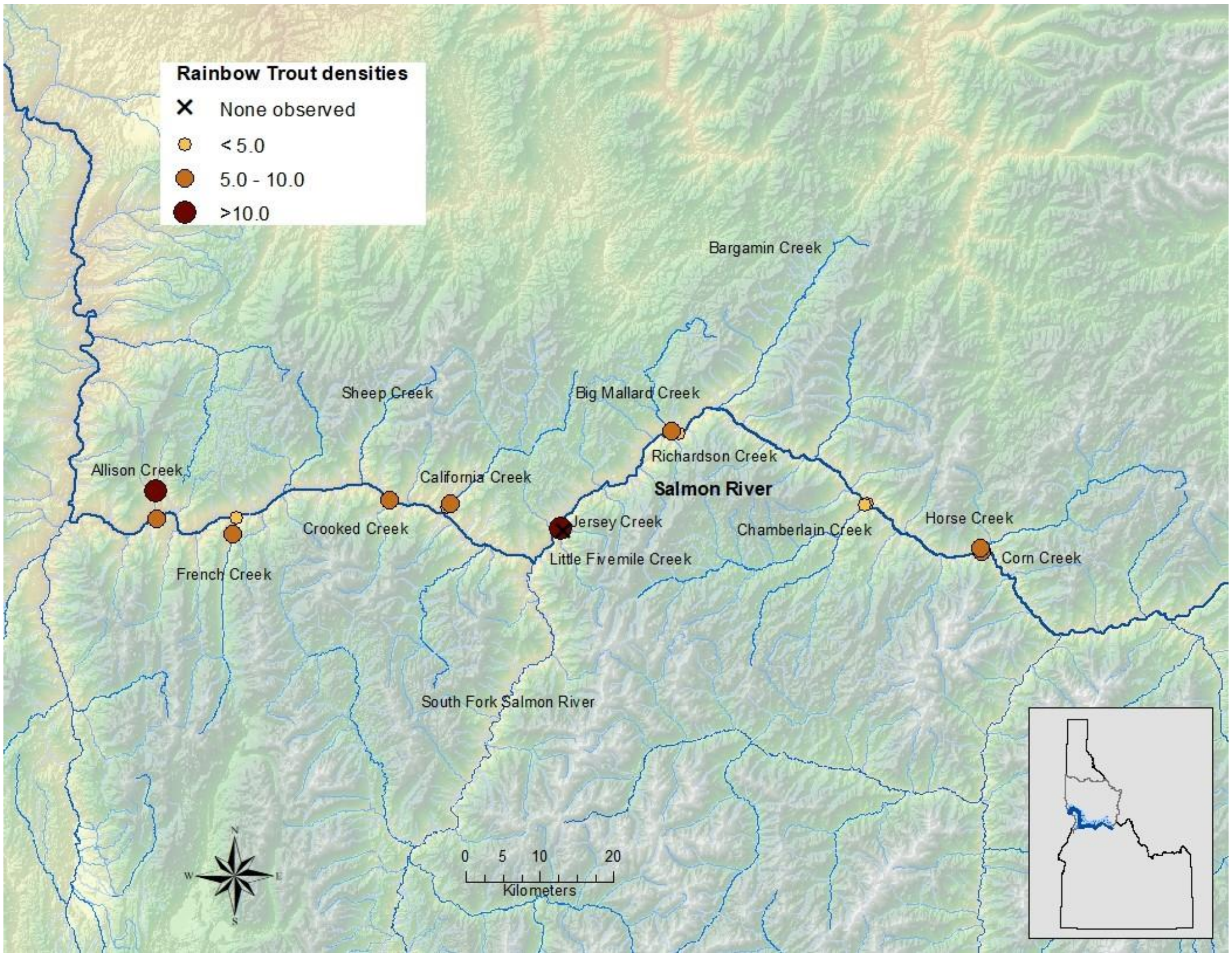

Figure 44. Areal density (fish/100 m^2) of Rainbow Trout observed in each snorkel transect surveyed in tributaries of the Salmon River, Idaho, during 2017.

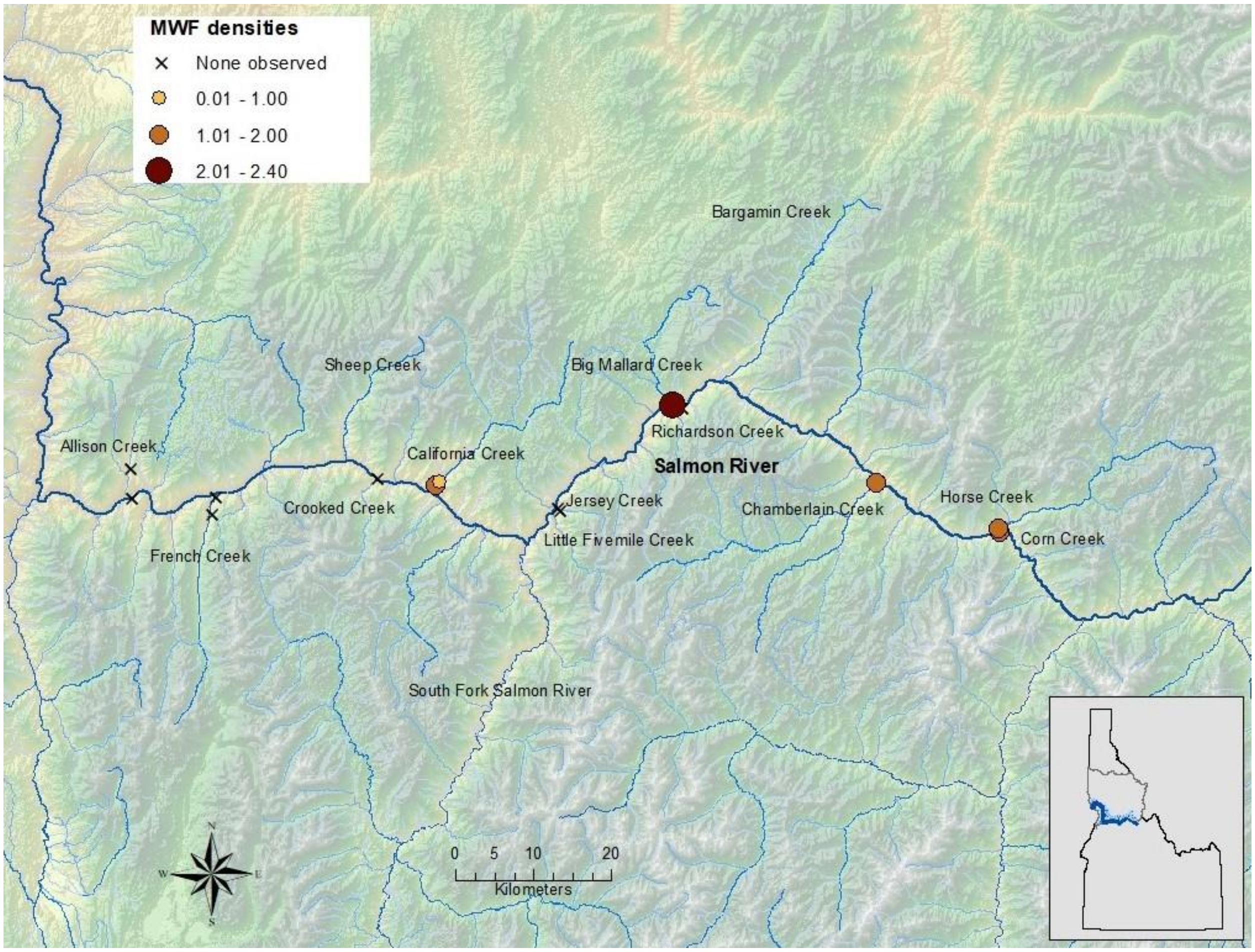

Figure 45. Areal density (fish/100 m^2) of Mountain Whitefish (MWF) observed in each snorkel transect surveyed in tributaries of the Salmon River, Idaho, during 2017.

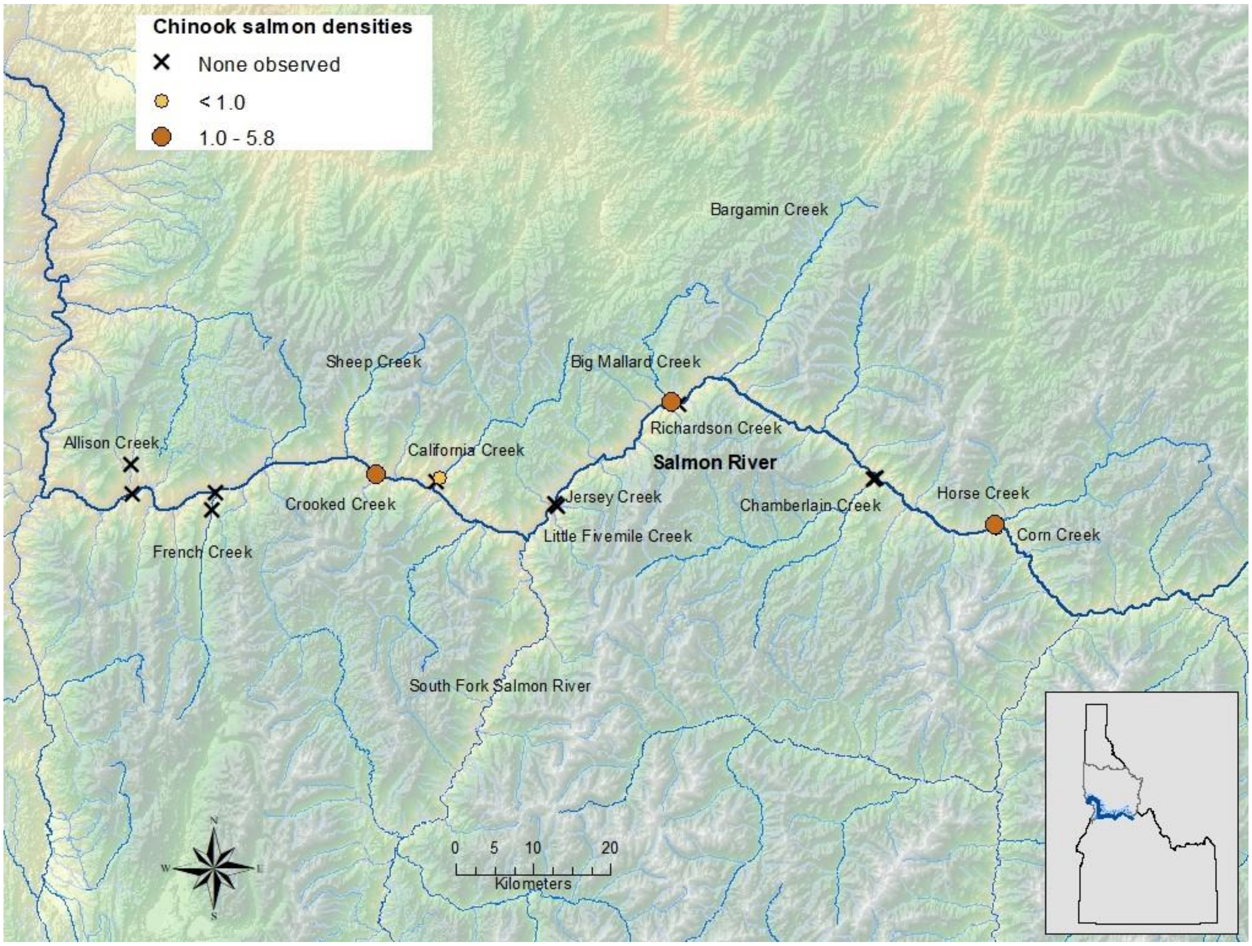

Figure 46. Areal density (fish/100 m^2) of Chinook salmon observed in each snorkel transect surveyed in tributaries of the Salmon River, Idaho, during 2017.

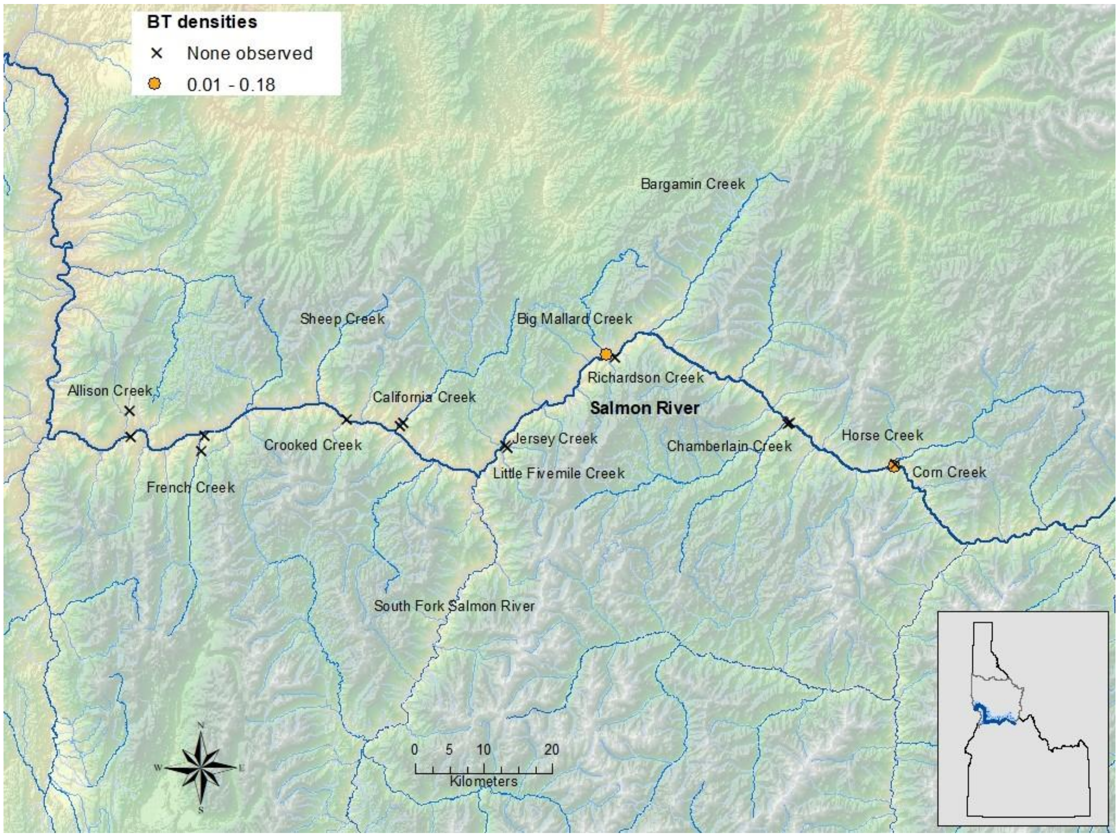

Figure 47. Areal density (fish/100 m^2) of Bull Trout (BT) observed in each snorkel transect surveyed in tributaries of the Salmon River, Idaho, during 2017.

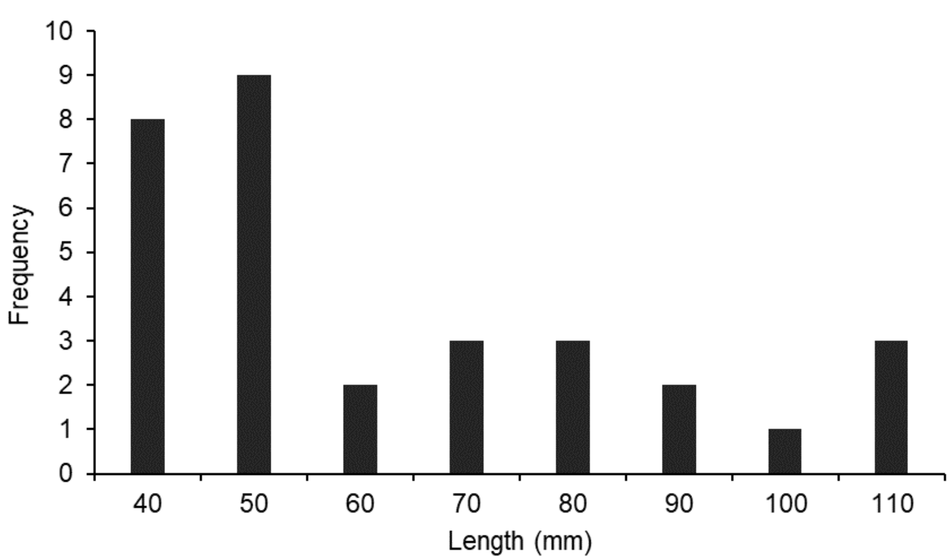

Figure 48. Length-frequency distribution of Pacific Lamprey (n = 31) sampled by backpack electrofishing in the Salmon River, Idaho, between Corn Creek and Allison Creek, in 2017.

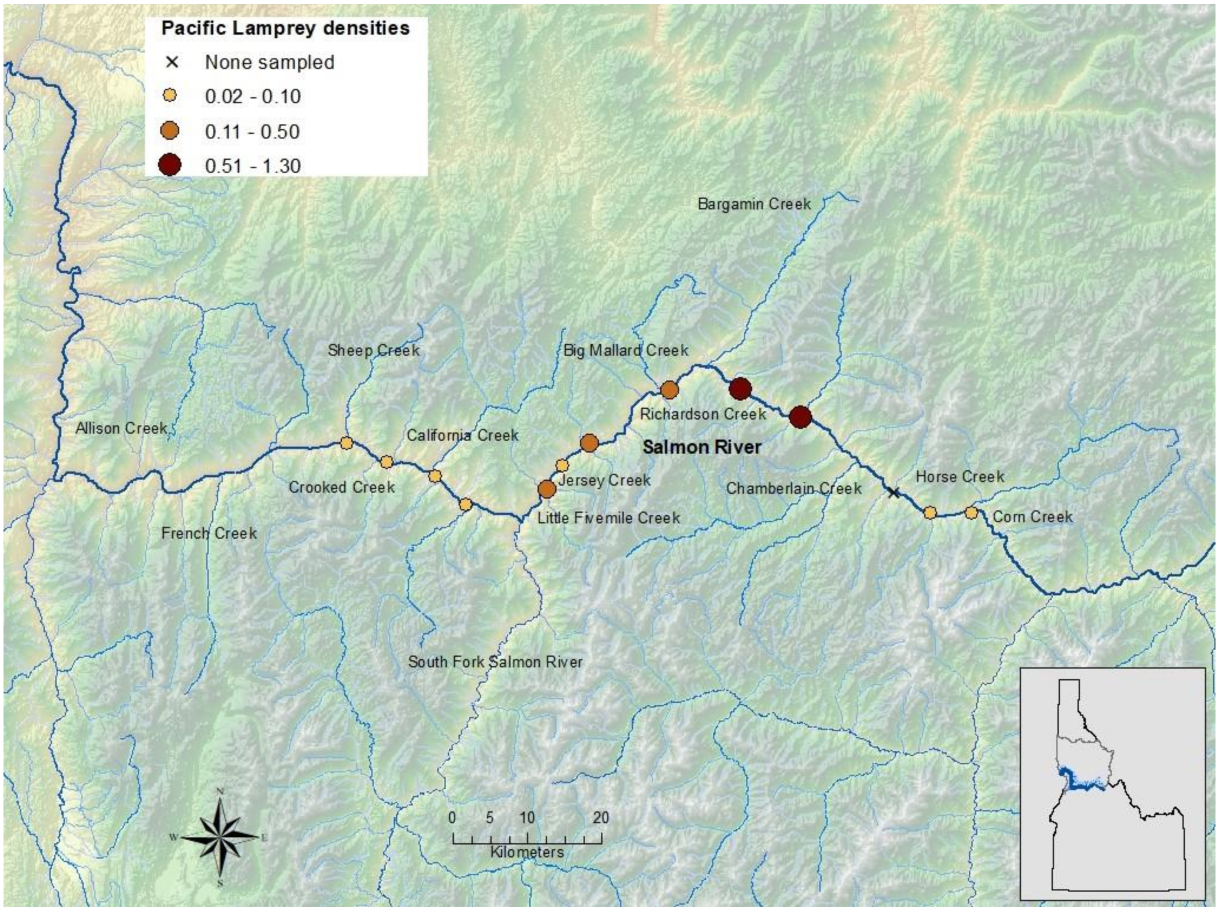

Figure 49. Linear density (fish/100 m transect length) of Pacific Lamprey sampled by backpack electrofishing in the Salmon River, Idaho, during 2017.

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EVALUATION OF FISH POPULATIONS IN THE SELWAY RIVER

ABSTRACT

Snorkel surveys were conducted in the Selway River drainage in 2017 to assess trends in Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT), Rainbow Trout *O. mykiss* (RBT), and Mountain Whitefish *Prosopium williamsoni* (MWF) density and size distribution. There has been a stable in WCT density over time (all fish and just those > 305 mm) in the main-stem and tributaries of the Selway River. There was a significantly declining trend in RBT density since 1988; however, there has been a stable trend in density since 1998. Since 1988, there has been a significant declining trend in MWF density in tributaries of the Selway River, but a significant increasing trend in the main-stem river. While hook-and-line catch rates of WCT have remained stable, there were increasing trends in both mean length and proportion of fish > 305 mm caught. Overall, our data indicates that the WCT population in the Selway River is stable or increasing in density and in proportion of fish > 305 mm, and is therefore meeting our management objectives. The decline in RBT density primarily occurred prior to 1990, when ~5,000 RBT were stocked annually in the Selway River. However, since RBT in the Selway River are likely primarily juvenile and residualized steelhead, the declining trends in adult steelhead returns to Idaho are likely having an impact as well. The primary drivers of MWF population trends in the Selway River are likely environmental factors affecting movement and recruitment, and potential survey biases that affect observability. In spite of the long-term increasing trend in the main-stem river, the recent downward trend in MWF densities across northern Idaho rivers and other parts of their historic range warrants a more thorough evaluation. Catch rates of WCT during the hook-and-line survey and the percentage of WCT > 305 mm are increasing, indicating we are meeting our management goal of providing a high quality fishery.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarkii lewisii* (WCT) are distributed throughout the Selway River drainage, occupying both the main river and tributaries. Both resident and fluvial life history forms are present. Fish populations in the Selway River have been regularly evaluated through snorkel surveys from White Cap Creek downstream to Selway Falls since 1973. Early studies of WCT in other northern Idaho rivers such as the St. Joe River, Kelly Creek, and the Lochsa River, concluded that the low WCT densities were a result of overfishing (Mallet 1967; Dunn 1968; Rankel 1971; Lindland 1977a). Concerns over declining WCT populations prompted IDFG to implement catch-and-release rules in the Selway River in 1976 (Lindland 1977b). Subsequent surveys showed that WCT density tripled over the four-year period after catch-and-release regulations were implemented. Similar trends in WCT density were also observed after catch-and-release regulations were implemented in the St. Joe River, Kelly Creek, and Lochsa River (Lindland 1977a).

After peaking in 1986, WCT counts have fluctuated, likely in response to drought, temperature extremes, flooding, and observer variability. Similar long-term fluctuations in WCT densities have also been observed in other Idaho rivers (Flinders et al. 2013; Ryan et al. 2014). As the majority of this watershed is afforded protected status through wilderness or roadless designations, land management and human development have little influence on WCT density. Limiting factors for WCT are therefore closely tied to natural regimes.

Due to limited vehicle access to this watershed, fishing pressure on the Selway River and its tributaries is relatively light. Currently, the fishery in the Selway watershed is managed under three different fishing rules. In all tributaries, a daily limit of two WCT is allowed. The rules on the main-stem Selway River for WCT are catch-and-release except for downstream of Selway Falls where a daily limit of two WCT > 356 mm is allowed from Memorial Day weekend to November 30. Downstream of Selway Falls also receives the most recreation as it is accessible by road. However, WCT use is limited in this reach of the Selway River during much of the summer due to unsuitable water temperatures. For these reasons, impacts from fishing are believed to have minimal influence on this WCT population. Monitoring this WCT population is important as it provides insight to trends in density in a watershed with light fishing effort, limited harvest, and little influence from land management activities. As demand on resident fisheries continues, it is important to track the status of this fish population to ensure continued quality fishing and to conserve wild native trout populations.

OBJECTIVES

1. Evaluate trends in density and size structure of WCT, Rainbow Trout *O. mykiss* (RBT) and Mountain Whitefish *Prosopium williamsoni* (MWF) in the Selway River on a two-out-of-three year basis.

STUDY AREA

The Selway River flows ~163 km from its headwaters in the Bitterroot Mountain Range to its confluence with the Lochsa River where it forms the Middle Fork Clearwater River (Figure 50). The Selway River watershed encompasses an area of ~5,200 km^2. The majority of the watershed occurs at elevations > 1,200 m. Land ownership in the Selway River watershed is almost 100% Federal and is managed by the U.S. Forest Service. About 95% of the watershed is afforded some level of protected status, primarily as wilderness (Selway Bitterroot and Frank Church River of No Return Wildernesses) or roadless areas. The Selway River has a road that parallels its path for the lower 30 km and about 15 km in the upper reaches.

METHODS

Field sampling

Snorkel survey

Westslope Cutthroat Trout, RBT, and MWF populations in the Selway River basin were surveyed through snorkeling 46 transects from July 19 to 26, 2017 (Figure 50). Transects in Moose Creek could not be snorkeled in 2017 due to wildfire-related closures. Two types of transects were snorkeled. The first group of transects are part of the General Parr Monitoring (GPM) program, developed to estimate anadromous fish response to Bonneville Power Administration habitat improvement projects (Scully et al. 1990). These transects are located on both the main-stem river and tributaries, and use standard snorkeling methodologies outlined in Apperson et al. (2015). This technique entails using an appropriate number of snorkelers to cover the entire width of the river to allow for the calculation of fish densities. They were conducted downstream in the main-stem river, and upstream in tributaries. The second group of surveys were 1-person transects developed for monitoring trends in density and size distribution of resident fish such as WCT and MWF*.* These transects are located on the main-stem river, and utilize one person starting at the upstream end of the transect and snorkeling downstream through the thalweg. Locations (GPS coordinates) and photographs of each 1-person snorkel transect are provided in Appendices “L” and “M” of DuPont et al. (2011). Of the 46 transects surveyed in 2017, 23 were historic GPM transects and 23 were 1-person transects.

For both types of snorkel surveys, all fish observed were counted, and length was estimated to the nearest inch for all game species. Other species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson, 50-m, reel-style, fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. The distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead *O. mykiss* and resident Rainbow Trout are indistinguishable, and are collectively referred to as “RBT”. This report focuses primarily on WCT, RBT, and MWF. Results and analysis of data collected on other species in 2017 can be found in Putnam et al. (2018).

Hook-and-line survey

Main-stem Selway River fish populations were surveyed by hook-and-line from July 19 to 26, 2017, while rafting from White Cap Creek to just upstream of Meadow Creek. Anglers utilized both fly and lure (spinners and spoons) techniques. Gear type, species, and total length (mm) were recorded for each fish captured. Anglers also noted any potential mortalities. Angler effort was recorded daily for each raft, but is estimated, as it is extremely difficult to calculate accurately due to the numerous interruptions that occur when rafting a technical river at low water, and the many stops required to conduct snorkel surveys.

Data Analysis

Snorkel survey

Analysis of 1-person transects was conducted on the available data as follows: all sizes of WCT, RBT, and MWF, and WCT > 305 mm (1973 - 2017); RBT and MWF > 305 mm (2002 - 2017). Analysis of GPM transects for WCT, RBT, and MWF was conducted on the available data as follows: main-stem (1992 - 2017); tributaries (1988 - 2017). We evaluated trends in WCT, RBT, and MWF density (fish/transect) in 1-person transects and density (fish/100 m^2) in GPM transects for all observed fish and just those > 305 mm using least squares regression. We used survey year as the independent variable and $\log_e$ transformed density as dependent variables (Maxell 1999; Kennedy and Meyer 2015). The length of time series available for each data set varied due to differences in when data collection started, and how data was collected/summarized. The intrinsic rate of change in the population (r_{intr}) is determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance, where the trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$. Distributions of WCT, RBT, and MWF were visually represented by plotting mean density for each transect on maps of the survey area using GIS software.

Hook-and-line survey

Analysis of hook-and-line surveys was conducted on the available data for WCT (1975 - 2017) and RBT (1997 - 2017). Rainbow Trout data was not recorded during these surveys until 1997. The relative density of fishes susceptible to hook-and-line fishing was assessed by calculating catch rates (fish/h) for all species combined and individual species. We also evaluated long-term trends in the number and size (mean length) of WCT and number of RBT caught through least squares regression as described above. Survey year was the independent variable and $\log_e$ transformed catch rates and mean length were dependent variables (Maxell 1999; Kennedy and Meyer 2015).

RESULTS

Snorkel survey

Westslope Cutthroat Trout

1-person transects

Mean WCT linear density in 1-person transects was 13.2/transect (Table 18). Westslope Cutthroat Trout were observed in all of these transects (Figure 51). The highest linear density of WCT in 1-person transects were observed between Moose Creek and Three-Links Creek (Table 19). Mean linear density of WCT in 2017 was higher than the long-term mean linear density (11.2/transect) for surveys conducted since 1973 (Figure 52); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in linear density from 1973 to 2017 (Table 20). The linear density of WCT > 305 mm has declined since 2013 (Figure 52); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in density of WCT > 305 mm from 1973 to 2017 (Table 20).

GPM main-stem transects

Mean WCT areal density in GPM main-stem transects was 0.48/100 m^2 (Table 18). Westslope Cutthroat Trout were observed in all of these transects. The highest areal densities of WCT were observed above White Cap Creek (Figure 51). Mean areal density of WCT in 2017 was lower than the long-term mean areal density (0.66/100 m^2) for surveys conducted since 1992 (Figure 53); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in areal density from 1992 to 2017 (Table 21).

GPM tributary transects

Mean WCT areal density in GPM tributary transects was 1.42/100 m^2 (Table 18). Westslope Cutthroat Trout were observed in every GPM tributary transect except one. The highest areal densities of WCT were observed in Marten and Deep creeks (Figure 51). Mean areal density of WCT in 2017 was lower than the long-term mean areal density (1.58/100 m^2) for surveys conducted since 1988 (Figure 53); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in areal density from 1988 to 2017 (Table 21).

Rainbow Trout

1-person transects

Mean RBT linear density in 1-person transects was 4.0/transect (Table 18). Rainbow Trout were observed in 57% of these transects. The highest linear density of WCT were observed in the Below Ham and Above Ladle transects (Figure 54). Mean linear density of RBT in 2017 was lower than the long-term mean linear density (9.6/transect) for surveys conducted since 1973 (Figure 55). Additionally, there was a statistically significant declining trend ($r_{intr} < 0$) in mean linear density from 1973 to 2017 (Table 20). The linear density of RBT > 305 mm in 2017 was 0.04/transect (Figure 55). Linear density of RBT > 305 mm has declined since 2013; however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in mean linear density of RBT > 305 mm from 2002 to 2017 (Table 20).

GPM main-stem transects

Mean RBT areal density in GPM main-stem transects was 0.13/100 m^2 (Table 18). Rainbow Trout were observed in 67% of these transects. The highest areal densities of RBT were observed above White Cap Creek (Figure 54). Mean areal density of RBT in 2017 was ~25% of the long-term mean areal density (0.53/100 m^2) for surveys conducted since 1992 (Figure 56). Additionally, there was a statistically significant declining trend ($r_{intr} < 0$) in RBT areal density in GPM main-stem transects from 1992 to 2017 (Table 21).

GPM tributary transects

Mean RBT areal density in GPM tributary transects was 2.50/100 m^2 (Table 18). Rainbow Trout were observed in every GPM tributary transect except one. The highest areal densities of RBT were observed in Marten and Three-Links creeks (Figure 54). Mean areal density of RBT in 2017 was lower than the long-term mean areal density (3.81/100 m^2) for surveys conducted since 1988 (Figure 56). Additionally, there was a statistically significant declining trend ($r_{intr} < 0$) in RBT areal density in GPM tributary transects from 1988 to 2017 (Table 21).

Mountain Whitefish

1-person transects

Mean MWF linear density in 1-person transects was 19.7/transect (Table 18). Mountain Whitefish were observed in all but two of these transects. The highest linear density of MWF was observed between Running Creek and Bear Creek (Figure 57). Mean linear density of MWF in 2017 was lower than the long-term mean linear density (24.3/transect) for surveys conducted since 1973 (Figure 58). Additionally, there was a statistically significant declining trend ($r_{intr} < 0$) in MWF linear density from 1973 to 2017 (Table 20). The linear density of MWF > 305 mm in 2017 was the highest since 2008 (Figure 58); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in linear density of MWF > 305 mm from 2002 to 2017 (Table 20).

GPM main-stem transects

Mean MWF areal density in GPM main-stem transects was 0.91/100 m^2 (Table 18). Mountain Whitefish were observed in all of these transects. The highest areal densities of MWF were observed between Running Creek and Bear Creek (Figure 57). Mean areal density of MWF in 2017 was lower than the long-term mean areal density (1.03/100 m^2) for surveys conducted since 1992 (Figure 59); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in MWF areal density in GPM main-stem transects from 1992 to 2017 (Table 21).

GPM tributary transects

Mean MWF areal density in GPM tributary transects was 0.69/100 m^2 (Table 18). Mountain Whitefish were observed in 64% of these transects. The highest areal densities of MWF were observed in Three-Links Creek (Figure 57). Mean areal density of MWF in 2017 was higher than the long-term mean areal density (0.62/100 m^2) for surveys conducted since 1988 (Figure 59); however, there was a statistically significant declining trend ($r_{intr} < 0$) in MWF areal density in GPM tributary transects from 1992 to 2017 (Table 21).

Hook-and-line survey

An estimated 135 angler hours resulted in the catch of 348 WCT, 40 RBT, 10 MWF, 1 WCT x RBT hybrids, and 1 Redside Shiner *Richardsonius balteatus*. The catch rate (for all species) of 3.0 fish/h in 2017 was close to the average for surveys conducted since 2012 (Table 22). The catch rate for WCT of 2.6/h was the highest of any survey conducted since 2012 (Table 22). Westslope Cutthroat Trout catch in 2017 was similar to the long-term average of 358 (Figure 60). Additionally, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in WCT catch during hook-and-line surveys conducted from 1975 to 2017 (Table 23). The mean length of WCT caught in 2017 (276 mm) was the highest since 2010, and was larger than the long-term average of 259 mm for hook-and-line surveys conducted from 1975 to 2017 (Figure 61). There was a statistically significant increasing trend ($r_{intr} > 0$) in mean length of WCT caught during hook-and-line surveys conducted from 1975 to 2017 (Table 23). The percent of WCT > 305 mm caught was the highest since 2010 (Figure 62). There was a statistically significant increasing trend ($r_{intr} > 0$) in percent of WCT > 305 mm caught during hook-and-line surveys conducted from 1975 to 2017 (Table 23).

The number of RBT caught by hook-and-line in 2017 was the lowest since 2010 (Figure 63). However, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in RBT catch during hook-and-line surveys conducted from 1997 to 2017 (Table 23).

DISCUSSION

Snorkel survey

Westslope Cutthroat Trout

Although density of WCT fluctuates from year to year, WCT density in the main-stem Selway River has been stable since 1973 (Figure 64). There has also been a stable trend in density of WCT > 305 mm during this time period. This contrasts with increasing population trends (total density and fish > 305 mm) observed in the North Fork Clearwater, St. Joe (SJR), and Coeur d'Alene (CDAR) rivers (Hand et al. 2020; Ryan et al. 2020). In the SJR and CDAR, increases were attributed to improvements in habitat, water quality, and favorable weather conditions (Ryan et al. 2014). From 1973 to 1986, WCT density increased fourfold to the highest density observed since 1973. This was likely in response to restrictive regulations implemented in 1976 (Lindland 1977b). Since then, densities have been lower, but have fluctuated around the long-term mean. With the Selway River located in a Wilderness area with limited access, minimal anthropogenic impacts, and restrictive regulations, environmental factors would likely be the driving force behind any changes that might occur in the WCT population as opposed to habitat and angler impacts.

Similar to the main-stem Selway River, WCT density in tributaries of the Selway River has been stable since 1988. Overall, it appears that the WCT population in the Selway River basin is stable for fish of all sizes and only those > 305 mm, and therefore meets our management objectives.

Rainbow Trout

There has been a significant declining trend in RBT density in the main-stem Selway River since 1973. This decline occurred prior to 2000, and may be partially explained by the cessation of stocking RBT in the Selway River in 1990. From 1968 to 1990, ~5,000 RBT were stocked into the Selway River annually. The density of RBT > 305 mm has remained low over time. A similar trend was observed in the Lochsa River (*See* Lochsa River section of this report). There was also a significant declining trend in RBT density in both the main-stem and tributaries of the Selway River. A similar trend was observed in the MFSR basin (Messner and Schoby 2019). Additionally, adult steelhead returns across Idaho have been declining (Dobos et al. 2020). Since many of the RBT observed in the Selway River are likely juvenile and residualized steelhead, the trends in RBT may be driven just as much by adult steelhead returns and juvenile survival as other factors. Since few RBT grow larger than 305 mm, they are not targeted heavily by anglers. Thus, angling and regulation changes have likely had little impact on trends in the RBT population in the Selway River.

Mountain Whitefish

There was a significant declining trend in MWF density in the main-stem Selway River since 1973. However, this decline has primarily occurred since 2003. In contrast to the main-stem Selway River, there was a significant declining trend in MWF density in tributaries prior to 2003.

Declines in MWF have been observed in other northern Idaho rivers, including the South Fork Clearwater River and Lochsa River (*See* SFCR and Lochsa River chapters in this report). Recently, declines in MWF have been documented in other populations across the southern portion of their range as well, including the Big Lost River and Kootenai River, Idaho, the Yampa River, Colorado, and the Madison River, Montana (Paragamian 2002; IDFG 2007; Boyer 2016). While the direct cause of these declines has not been identified, these declines have been linked to occurrences of low flows and higher water temperatures (Brinkman et al. 2013). These studies also suggested that habitat alteration, irrigation, nonnative fish interactions, disease, and harvest are also likely contributing to declines in MWF populations (IDFG 2007; Boyer 2016). Some of these factors generally do not apply to the wilderness of the Selway River drainage; however, increases in water temperature, disease, and survey timing could be potential factors.

Increased water temperatures could affect MWF populations in several ways. Warmer temperatures may impact populations by moving more fish out of the main-stem and larger tributaries (where we surveyed) and/or through increased mortality (Hunt 1992; Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Mean monthly summer air temperatures have been above normal every year except one since 1996 (NOAA 2021). Additionally, severe outbreaks of Proliferative Kidney Disease (PKD) have been observed in Montana, and the disease is known to be present in Idaho (Phillips 2016; Hutchins et al. 2021). While no major fish kills have been directly observed, minor die-offs have been observed in in Idaho rivers during summer months. Thus, PKD could impact populations through lower level mortality. As such, increased water temperatures could impact MWF movement, survival, and recruitment before some other species.

Sampling biases may also be playing a part in the recent decline in main-stem density. In 2005, we began conducting our snorkel surveys based on river flow instead of a set calendar date. Surveys now tend to occur at higher flow levels, which could reduce visibility of a bottom-dwelling fish compared to previous surveys. This fits with the trends in density observed before and after 2003.

The trends observed in MWF density and density in the Selway River are likely a combination of changes in our sampling strategy and potential disease/environmental factors affecting adult survival and recruitment. Regardless, the recent downward trend in MWF densities across northern Idaho rivers and other parts of their historic range warrants a more thorough evaluation.

Hook-and-line survey

There has been an increasing trend in WCT average length and percent of fish > 305 mm caught by hook-and-line for surveys conducted since 1973. The percent of WCT > 305 mm caught is similar to those reported for Middle Fork Salmon River (MFSR) float trips (Messner and Schoby 2019). In contrast to the increasing trend in the Selway River, this proportion has remained stable in the MFSR. The catch rate of 3.0 fish/h for hook-and-line surveys was similar to the average for surveys conducted in the Selway River from 2012 to 2017, but at the lower end of the range (2.8 - 5.8) of catch rates on float trips conducted on the Middle Fork Salmon River (Messner and Schoby 2019). However, the catch rate for WCT (2.6/h) was similar to the Middle Fork Salmon River in 2016 and 2017 (Messner and Schoby 2019). The stable annual catch, and long-term increasing trends in WCT length and percent > 305 mm observed in the Selway River indicate we are meeting our management goals of providing a high-quality fishery with abundant larger fish.

Annual catch of RBT during float trips on the Selway River has remained stable. This aligns with the lack of a trend in RBT density in 1-person snorkel transects during similar time periods (mid-1990's to present). As discussed previously, this is likely attributable to the cessation of RBT stocking in the Selway River in 1990.

MANAGEMENT RECOMMENDATIONS

1. Continue to conduct Selway River snorkel and hook-and-line surveys to monitor trends in WCT, RBT, and MWF density and size structure.

Table 18. Fish density in GPM transects (fish/100 m^2) and 1-person transects (fish/transect), by transect, for snorkel surveys of the Selway River drainage, Idaho, in 2017.

GPM transects

River section	Transect name	Transect length (m)	Transect width (m)	Temp °C	Density: Westslope Cutthroat Trout	RBT	Mountain Whitefish	Chinook Salmon	Trout fry	Bull Trout
Tributaries	Bear Creek Lower	54	18	14.0	0.62	2.80	0.52	0.60	0.00	0.00
	Bear Creek Upper	116	21	14.0	0.00	1.00	0.12	0.00	0.00	0.00
	Deep Cr, Cactus	70	8	14.0	1.04	3.70	0.00	0.30	0.90	0.00
	Deep Cr, Scimitar	80	7	14.0	2.39	1.70	0.00	1.70	3.70	0.00
	Little Clearwater, #1	38	11	15.5	1.23	0.00	0.49	1.50	0.00	0.20
	Little Clearwater, #2	50	13	16.0	0.47	1.90	0.47	0.20	0.20	0.00
	Marten	70	7	12.0	11.11	10.80	0.00	6.80	0.00	0.00
	Moose Creek 1	51	36	20.0	0.25	0.60	1.25	0.00	0.30	0.00
	Moose Creek, East Fork #2	-	-	-	-	-	-	-	-	-
	Moose Creek, East Fork #3	-	-	-	-	-	-	-	-	-
	Moose Creek, North Fork	-	-	-	-	-	-	-	-	-
	Running Creek 1	42	14	20.5	0.62	0.50	0.00	0.50	0.70	0.00
	Running Creek 2	68	12	20.0	0.38	3.50	0.00	0.00	6.00	0.00
	Three Links 1	31	9	15.0	0.74	4.80	5.50	0.00	2.60	0.00
	White Cap, Strata 3, #1	82	16	20.0	0.39	2.40	1.02	0.20	0.20	0.00
	White Cap, Strata 3, #2	98	19	17.5	0.33	1.00	0.11	1.30	0.50	0.00
	White Cap, Strata 3, #3	104	15	16.5	0.25	0.30	0.19	0.90	1.90	0.00
Above White Cap Cr.	Hell's Half Acre	79	15	12.0	0.20	0.00	1.50	0.00	0.10	0.00
	Magruder Crossing	152	24	14.5	0.38	0.14	0.16	0.58	0.00	0.00
	Beaver Point	152	16	17.0	0.41	0.58	0.25	3.87	0.62	0.04
	Little Clearwater	70	18	16.0	1.03	0.32	1.75	2.78	0.32	0.16
Running Cr. to Bear Cr.	Badluck Cr	77	41	20.0	0.86	0.06	1.43	0.00	0.00	0.03
	Big Bend	108	37	16.0	0.15	0.00	0.53	0.00	0.83	0.00
	Northstar	94	36	18.0	0.35	0.03	1.68	0.03	0.68	0.00
Bear Cr. to Moose Cr.	Osprey Island	120	40	17.0	0.67	0.06	0.79	0.02	0.23	0.00
Moose Cr. to Three-links Cr.	Below Tango	144	47	17.0	0.24	0.00	0.09	0.00	0.00	0.00
	Average				1.05	1.57	0.78	0.92	0.86	0.02
	90% CI				0.77	0.84	0.41	0.56	0.50	0.02

1-person transects

River section	Transect name	Transect length (m)	Transect width (m)	Temp °C	Abundance: Westslope Cutthroat Trout	RBT	Mountain Whitefish	Chinook Salmon	Trout fry	Bull Trout
White Cap Cr. to Runnning Cr.	1/2 mile below white cap	62	15	-	2	3	27	4	0	0
	1 mile below white cap	81	31	-	7	0	36	0	0	0
	cougar bluff	66	16	-	10	14	20	0	0	0
Running Cr. to Bear Cr.	1/2 mile below running	54	27	-	15	3	50	0	0	0
	archer	52	25	-	4	0	48	0	0	0
	above goat creek rapid	94	30	-	34	2	30	0	0	0
	selway lodge	75	23	-	7	1	24	0	7	0
Bear Cr. to Moose Cr.	above rodeo	54	23	-	1	0	7	0	0	0
	below rodeo	95	15	-	18	0	3	0	0	0
	below pettibone	80	32	-	4	0	4	0	0	0
	rattlesnake bar	130	42	-	3	1	5	0	0	0
	below ham	110	25	-	6	31	0	0	0	0
	below hell creek	-	-	-	-	-	-	-	-	-
Moose Cr. to Three-Links Cr.	moose creek confluence	55	28	-	5	3	4	0	0	0
	divide creek	100	31	-	43	2	25	0	0	1
	above ladle	120	35	-	2	25	4	0	0	0
	below ladle	85	31	-	26	0	4	0	0	0
	below osprey rapid	90	24	-	22	1	1	0	0	0
Three-Links Cr. to Race Cr.	below 3-links	150	31	-	8	0	1	0	0	0
	dry bar	50	28	-	34	1	150	0	0	0
	above wolf creek	150	47	-	9	0	5	0	0	0
	above renshaw	80	51	-	40	0	3	0	0	0
	Otter	25	10	-	1	0	0	0	0	0
	Packer	65	42	-	2	5	2	0	0	0
	Average				13.2	4.0	19.7	0.2	0.3	0.0
	90% CI				4.6	1.4	6.9	0.1	0.1	0.0

Table 19. Average density (fish/transect) of all sizes of Westslope Cutthroat Trout (WCT) and just those > 305 mm in sections of the main-stem Selway River, Idaho, determined by 1-person snorkel surveys from 1973 to 2017.

All WCT

	River section					
Year	White Cap Creek to Running Creek	Running Creek to Bear Creek	Bear Creek to Moose Creek	Moose Creek to Three-Links Creek	Three Links Creek Race Creek	Mean
1973	4.2	7.2	5.3	4.5	5.0	5.2
1974	3.4	4.8	7.5	8.2	3.4	5.5
1975	6.8	6.6	5.0	6.3	4.6	5.9
1976	7.2	6.2	6.0	8.8	6.1	6.9
1977	10.8	18.6	17.4	22.0	9.3	15.6
1978	7.4	10.6	19.6	20.9	9.8	13.6
1980	13.2	18.6	16.0	21.7	17.2	17.3
1982	11.2	11.2	16.2	20.3	20.8	15.9
1984	11.0	17.4	19.4	25.7	16.3	17.9
1986	15.2	19.2	21.4	26.1	24.6	21.3
1988	13.3	11.6	21.8	24.3	17.4	17.7
1990	6.8	16.4	7.4	6.8	11.7	9.8
1992	4.8	9.4	6.2	4.4	3.0	5.6
1994	7.5	9.0	8.3	3.0	6.0	6.8
1995	13.0	13.3	13.3	6.0	6.4	10.4
1996	10.7	15.5	15.0	8.5	30.0	15.9
1997	6.0	26.5	7.8	10.5	15.0	13.2
1998	---	---	1.0	2.0	7.6	---
1999	17.0	12.6	16.6	10.6	4.2	12.2
2001	13.3	12.7	7.5	5.3	1.3	6.2
2002	12.7	21.0	8.6	12.6	2.2	9.8
2003	10.3	8.3	10.6	---	---	---
2004	8.0	5.0	7.0	12.0	5.5	5.6
2005	13.5	6.0	8.4	19.8	6.7	10.1
2007	2.3	4.5	3.6	1.8	15.3	6.3
2008	15.3	8.5	15.0	14.8	10.3	10.6
2009	6.7	4.0	10.2	21.4	12.5	11.2
2010	7.0	9.0	13.8	31.3	16.0	13.1
2011	11.5	10.2	12.5	22.8	11.0	10.0
2012	5.3	8.8	5.8	9.2	8.8	7.8
2013	4.7	16.2	8.5	52.8	11.2	15.0
2015	8.0	12.0	17.2	13.3	10.6	12.8
2017	6.3	15.0	6.4	19.6	15.7	13.2

Table 19 (continued)

WCT > 305

	River section					
Year	White Cap Creek to Running Creek	Running Creek to Bear Creek	Bear Creek to Moose Creek	Moose Creek to Three-Links Creek	Three Links Creek Race Creek	Mean
1973	0.4	0.8	1.8	0.6	1.2	1.0
1974	0.6	0.4	1.2	1.3	0.3	0.7
1975	0.8	1.2	0.4	0.7	1.4	0.9
1976	1.6	1.0	1.5	1.9	1.2	1.4
1977	2.4	4.0	4.4	3.3	2.5	3.3
1978	1.2	2.2	4.2	3.1	3.0	2.7
1980	1.7	2.2	1.6	3.9	1.8	2.2
1982	1.0	1.2	2.4	4.2	3.5	2.5
1984	1.7	3.6	4.4	6.2	4.8	4.1
1986	3.2	2.8	4.0	5.9	3.6	3.9
1988	3.3	2.6	5.0	5.8	3.2	4.0
1990	2.0	2.6	1.2	1.4	3.7	2.2
1992	0.3	2.4	3.0	0.3	0.9	1.4
1994	0.5	1.0	1.0	0.0	0.0	0.5
1995	0.0	0.0	1.2	0.3	0.0	0.3
1996	0.0	0.0	0.0	0.9	3.0	0.8
1997	0.5	3.0	1.5	1.6	2.3	1.8
1998	---	---	0.0	0.3	3.0	---
1999	1.0	0.3	2.8	3.4	1.0	1.7
2001	3.7	5.0	4.2	2.3	0.7	6.2
2002	5.7	4.3	2.3	2.2	0.8	9.8
2003	0.7	1.7	3.3	---	---	---
2004	1.0	1.0	1.4	1.7	0.5	5.6
2005	4.0	1.0	0.9	3.4	1.8	10.1
2007	0.3	2.0	0.7	0.0	5.0	6.3
2008	2.7	1.8	2.0	0.8	0.8	10.6
2009	0.0	0.3	0.8	2.8	4.3	11.2
2010	1.7	2.8	2.0	2.3	4.0	13.1
2011	3.8	4.7	3.8	7.0	1.3	10.0
2012	0.3	0.3	0.2	3.0	2.2	7.8
2013	3.0	7.8	2.0	2.8	1.8	15.0
2015	1.0	0.8	2.7	2.3	3.8	12.8
2017	1.0	0.8	1.2	3.0	0.3	13.2

Table 20. Intrinsic rate of change (r_{intr}) in density (fish/transect) for Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and Mountain Whitefish (MWF) observed in 1-person snorkel surveys conducted in the main-stem Selway River, Idaho, from 1973 to 2017. Significance was set at α = 0.10. Data sets include fish of all sizes unless specified as "> 305 mm", which only includes fish of this size.

Species	Data set	r_{intr} estimate	90% CI lower	90% CI upper
WCT	All fish	0.004	-0.006	0.014
	> 305 mm	0.010	-0.008	0.028
RBT	All fish			
	1973-2017	-0.054	-0.071	-0.037
	> 305 mm			
	2002-2017	0.016	-0.002	0.032
MWF	All fish			
	1973-2017	-0.011	-0.021	-0.002
	1973-2003	-0.002	-0.019	0.015
	2003-2017	-0.089	-0.139	-0.040
	> 305 mm			
	2002-2017	-0.050	-0.121	0.021

Table 21. Intrinsic rate of change (r_{intr}) in density (fish/100 m^2) for Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and Mountain Whitefish (MWF) observed in snorkel surveys conducted in the main-stem (1992 - 2017) and tributaries (1988 - 2017) of the Selway River, Idaho. Significance was set at α = 0.10.

Species	Data set	r_{intr} estimate	90% CI lower	90% CI upper
WCT	Tributaries	0.020	-0.004	0.043
	Main-stem	0.004	-0.017	0.025
RBT	Tributaries	-0.034	-0.059	-0.009
	Main-stem	-0.063	-0.103	-0.024
MWF	Tributaries	-0.029	-0.049	-0.008
	1988-2003	-0.081	-0.135	-0.027
	2003-2017	0.028	-0.031	0.088
	Main-stem	0.012	-0.009	0.032
	1992-2003	0.073	-0.005	0.152
	2003-2017	-0.053	-0.082	-0.025

Table 22. Effort (h) and catch rates (fish/h; all species combined and only Westslope Cutthroat Trout) for hook-and-line surveys of the Selway River, Idaho, from 2012 to 2017.

		Catch rate	
Year	Effort	Overall	WCT
2012	155	3.4	2.5
2013	140	2.7	2.2
2015	140	3.3	2.5
2017	135	3.0	2.6

Table 23. Intrinsic rate of change (r_{intr}) in the number of fish caught (catch), mean length, and proportion of fish > 305 mm for Westslope Cutthroat Trout (WCT; 1975 - 2017) and Rainbow Trout (RBT; 1997 - 2017) caught during hook-and-line surveys in the Selway River, Idaho. Significance was set at α = 0.10.

		r_{intr}	90% CI	
Species	Data set	estimate	lower	upper
WCT	Catch	-0.004	-0.013	0.006
	Mean length	0.002	0.001	0.002
	> 305 mm	0.016	0.010	0.021
RBT	Catch	0.003	-0.091	0.096

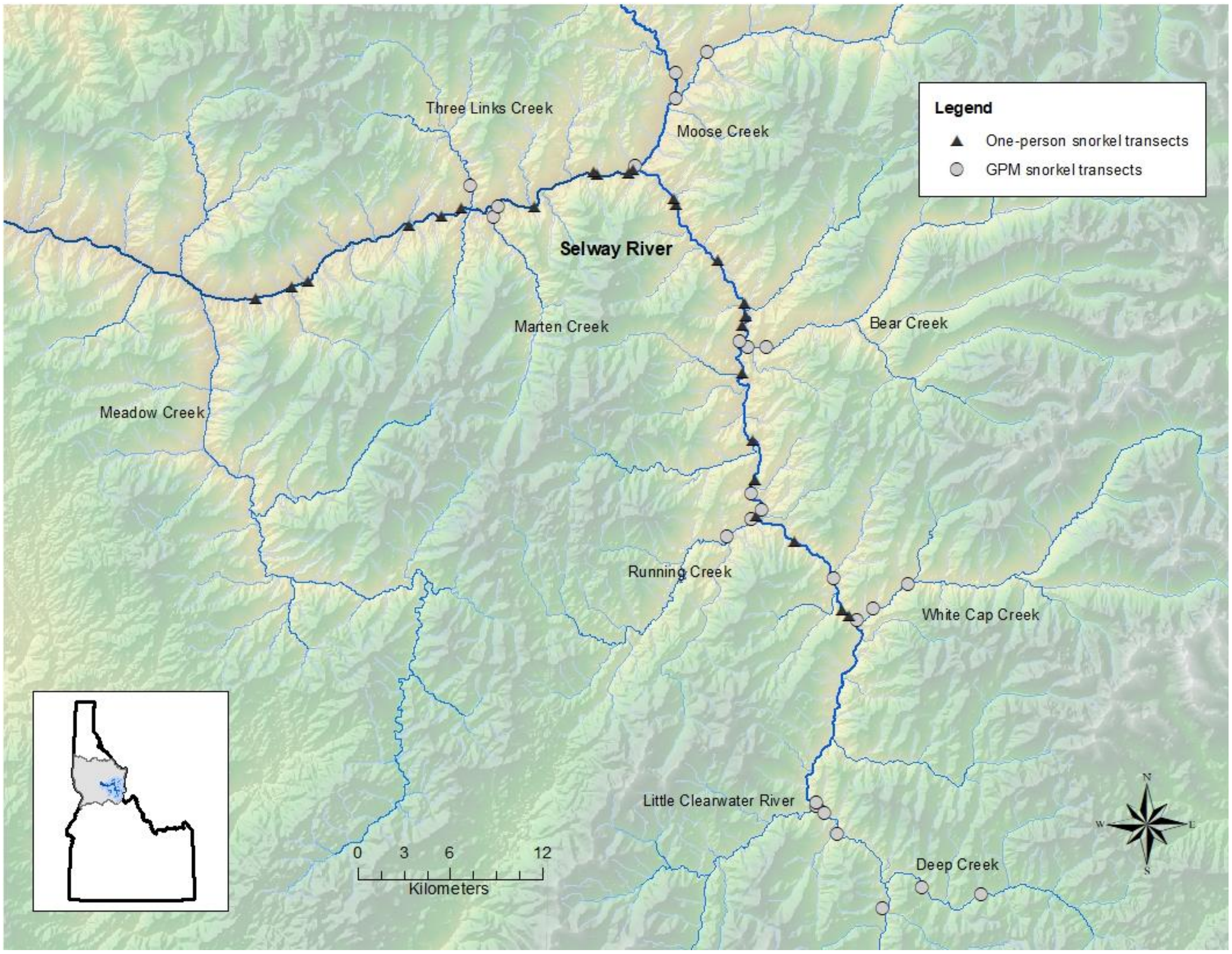

Figure 50. Map showing locations of General Parr Monitoring (GPM) and 1-person snorkel transects surveyed in the Selway River basin, Idaho, in 2017.

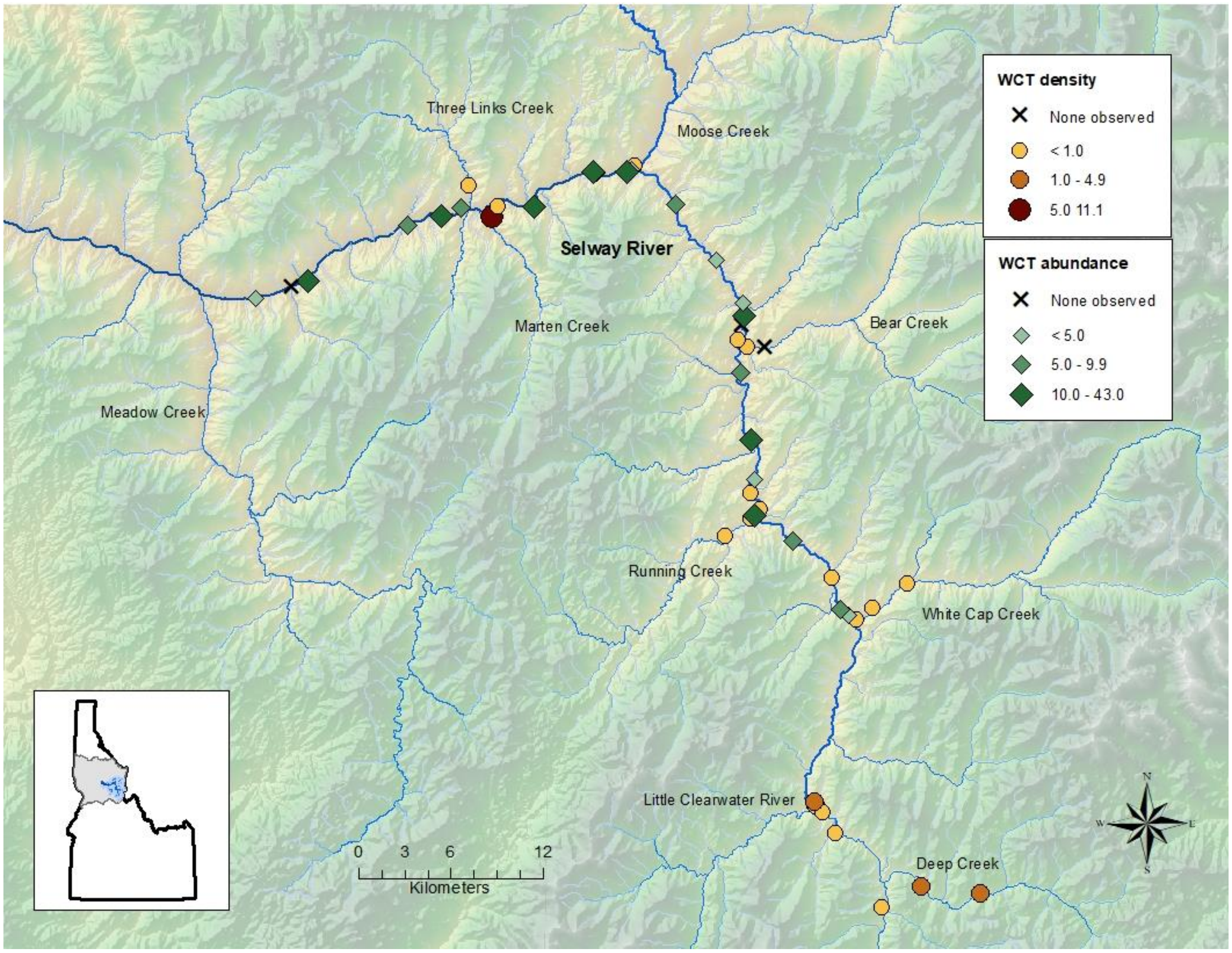

Figure 51. Westslope Cutthroat Trout density (fish/100 m^2) in General Parr Monitoring transects and density (fish/transect) in 1-person transects observed for each snorkel transect surveyed in the Selway River basin, Idaho, in 2017.

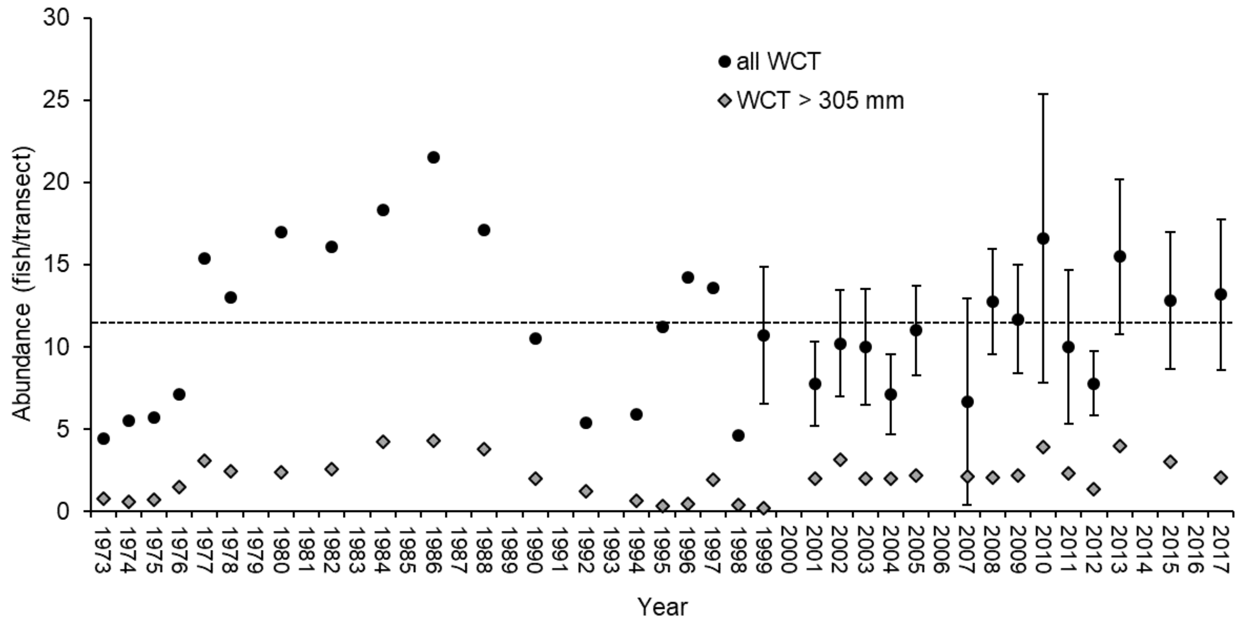

Figure 52. Mean density (all sizes and only fish > 305 mm) of Westslope Cutthroat Trout observed in 1-person snorkel transects in tributaries and the main-stem Selway River, Idaho, from 1973 to 2017. Dashed line indicates mean density for all sizes of WCT. Error bars represent 90% confidence intervals.

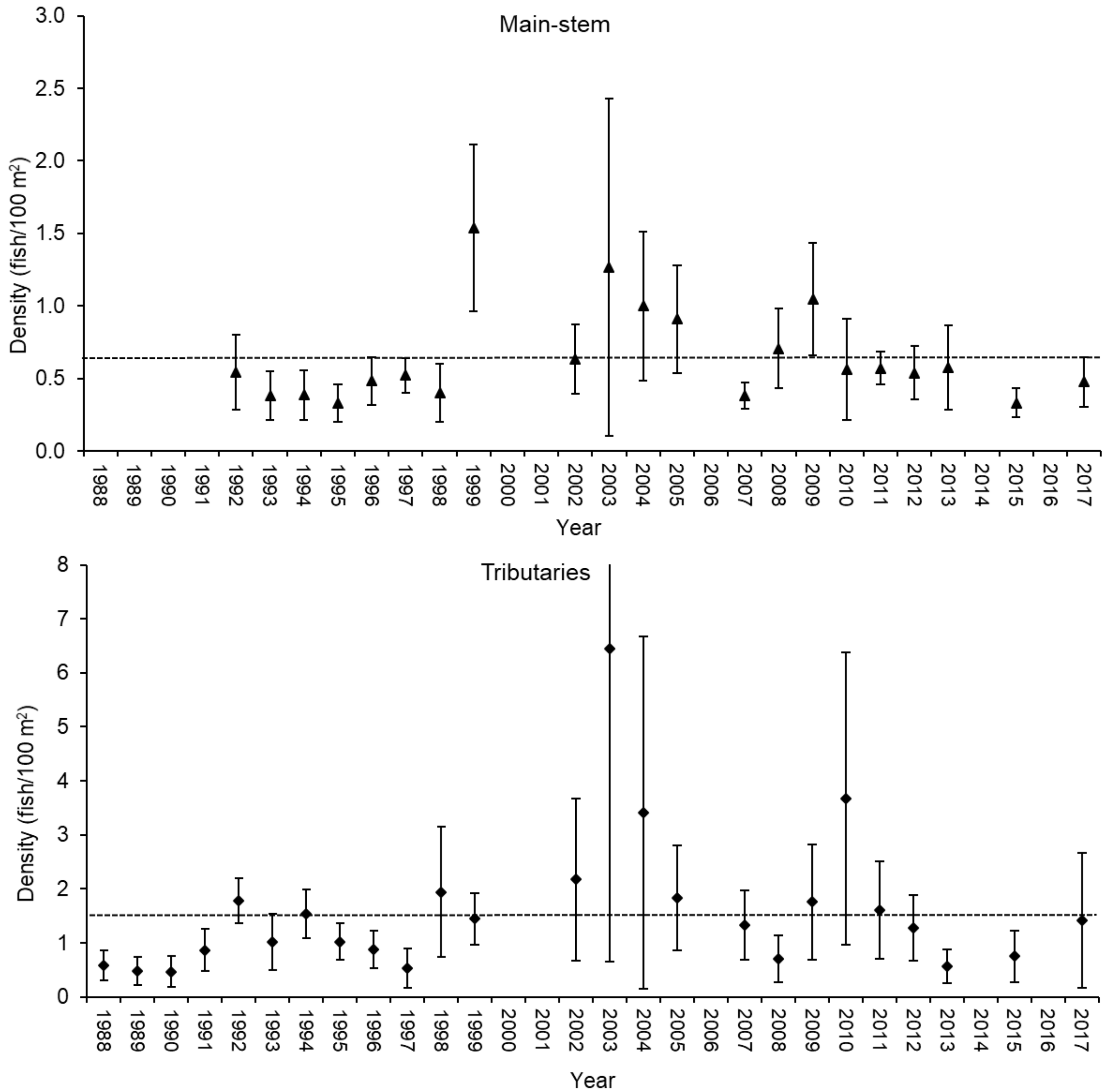

Figure 53. Mean densities of Westslope Cutthroat Trout observed in General Parr Monitoring snorkel transects in the main-stem (1992 - 2017) and tributaries (1988 - 2017) of the Selway River, Idaho. Dashed lines indicate mean densities of WCT. Error bars represent 90% confidence intervals.

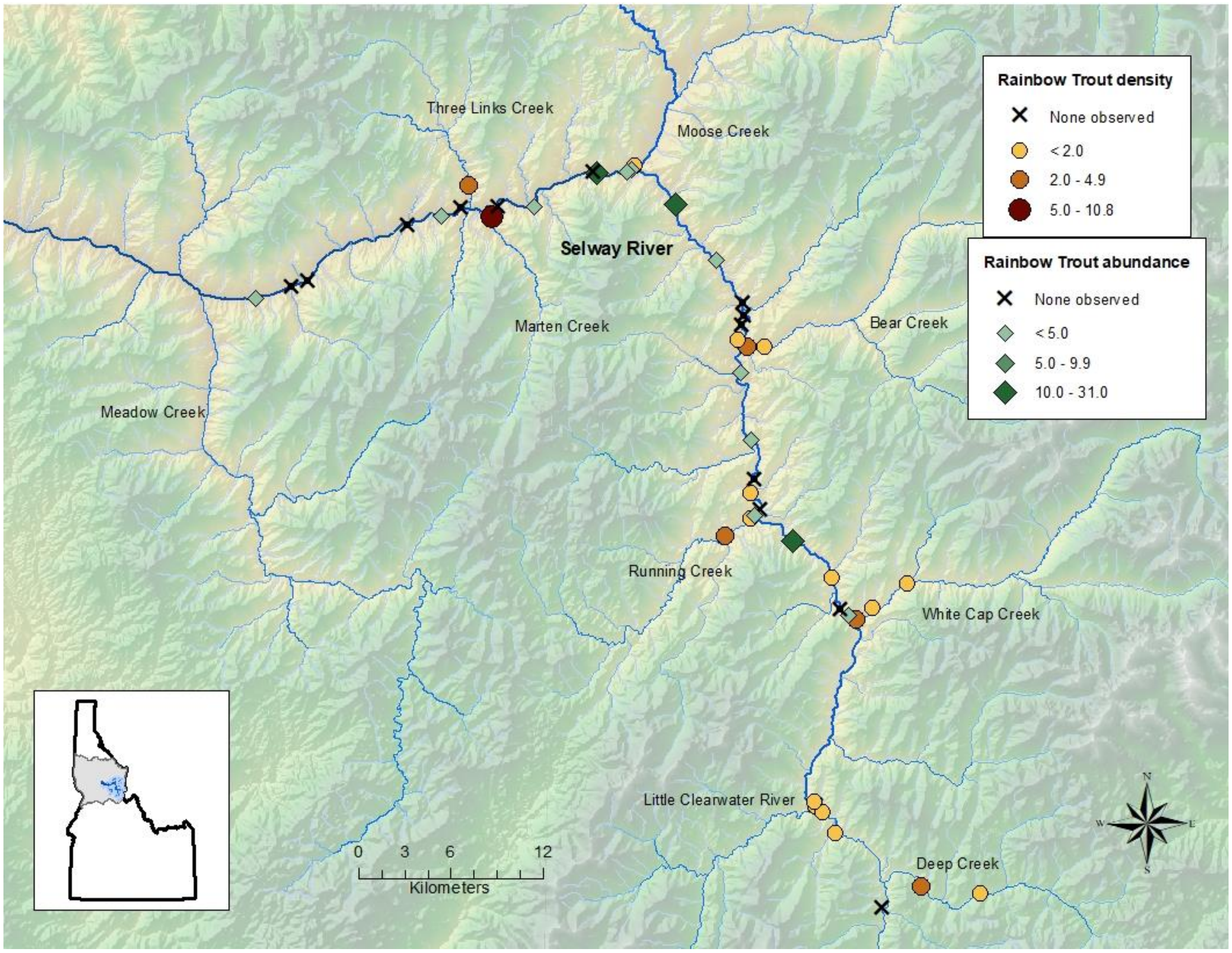

Figure 54. Rainbow trout density (fish/100 m^2) in General Parr Monitoring transects and density (fish/transect) in 1-person transects observed for each snorkel transect surveyed in the Selway River basin, Idaho, in 2017.

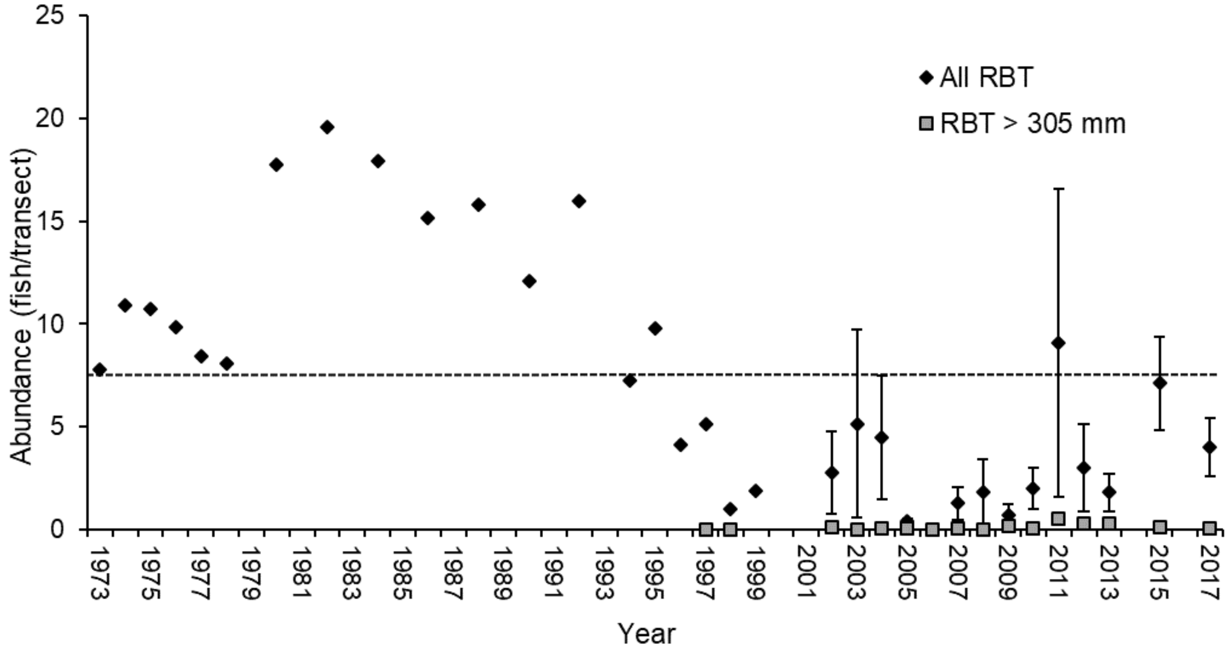

Figure 55. Mean density of Rainbow Trout observed in 1-person snorkel transects in the main-stem Selway River, Idaho, from 1973 to 2017. Dashed line indicates mean density for all sizes of Rainbow Trout. Error bars represent 90% confidence intervals.

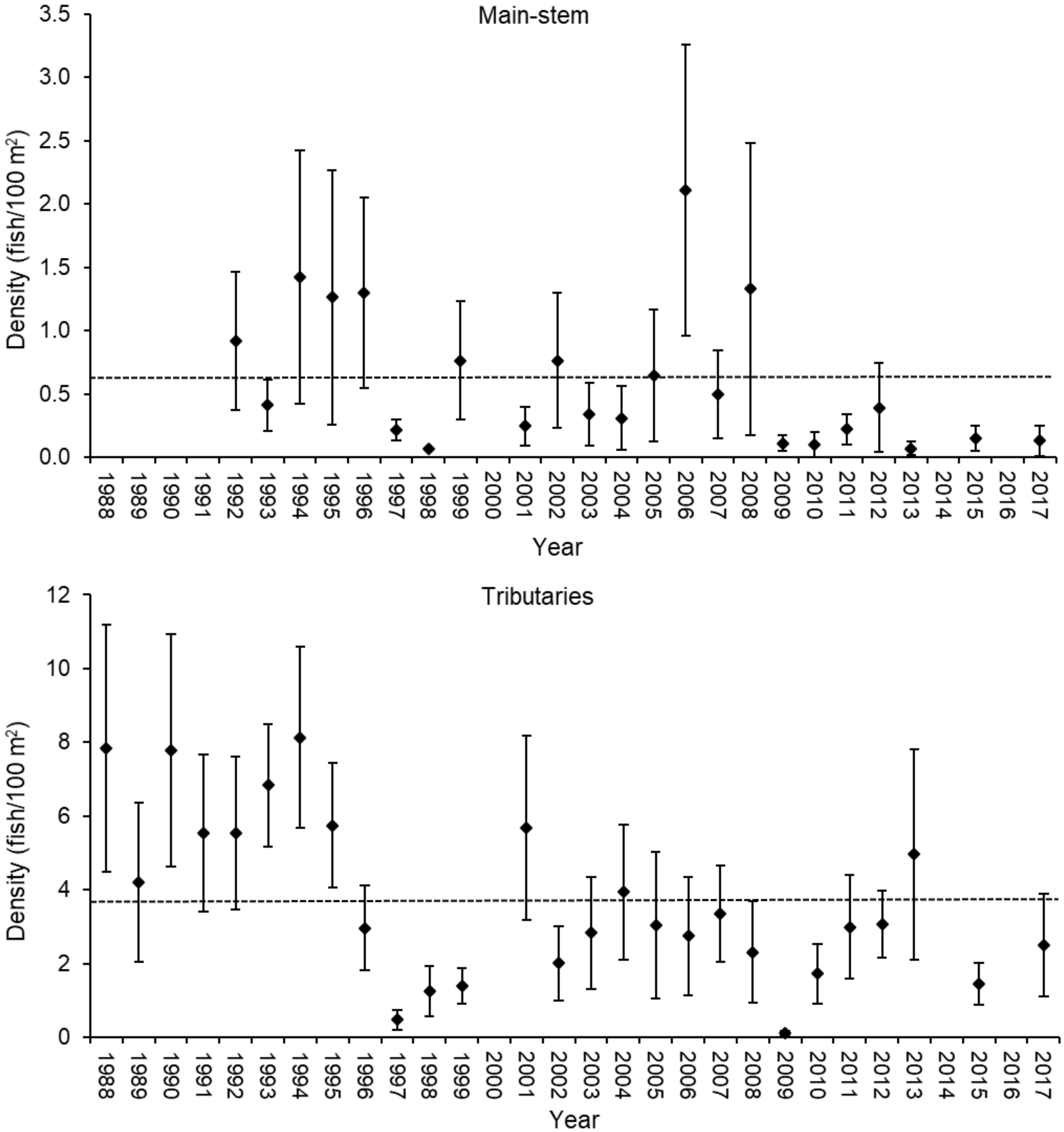

Figure 56. Mean density of Rainbow Trout observed in General Parr Monitoring snorkel transects in the main-stem (1992 - 2017) and tributaries (1988 - 2017) of the Selway River drainage, Idaho, from 1988 to 2017. Dashed lines indicate mean densities. Error bars represent 90% confidence intervals.

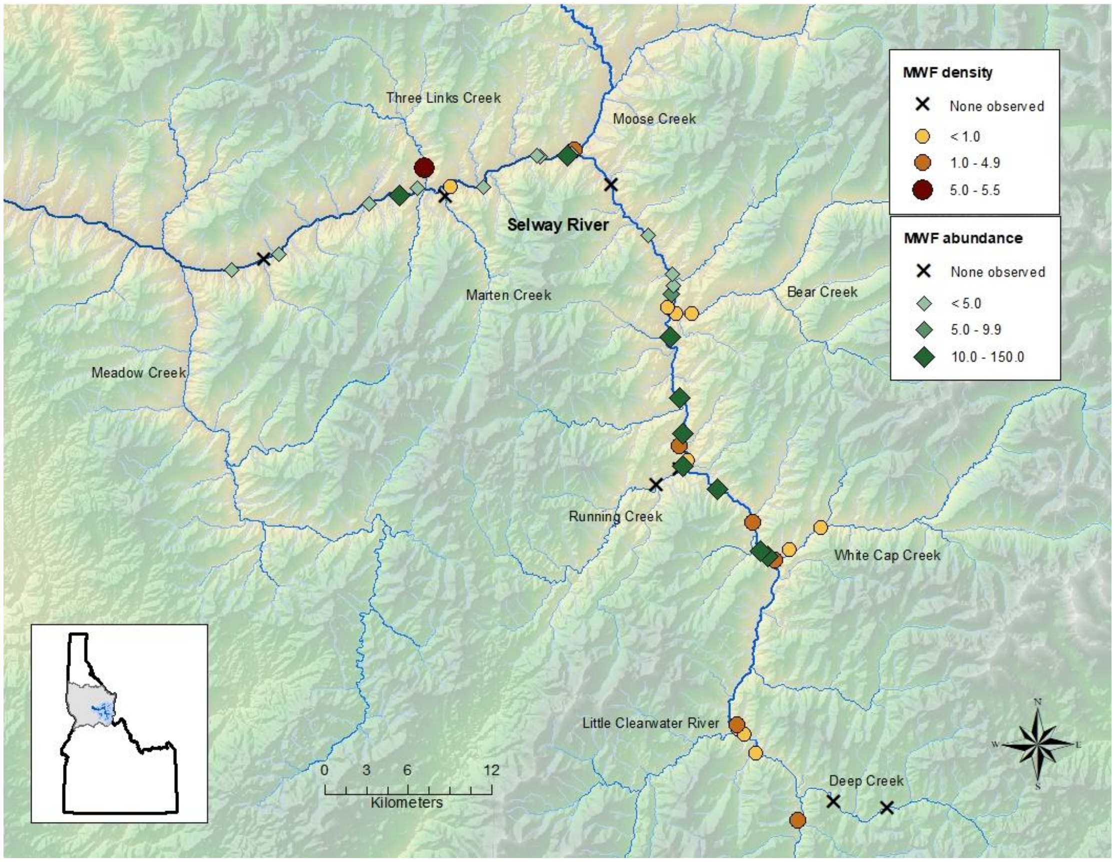

Figure 57. Mountain Whitefish density (fish/100 m^2) in General Parr Monitoring transects and density (fish/transect) in 1-person transects observed for each snorkel transect surveyed in the Selway River basin, Idaho, in 2017.

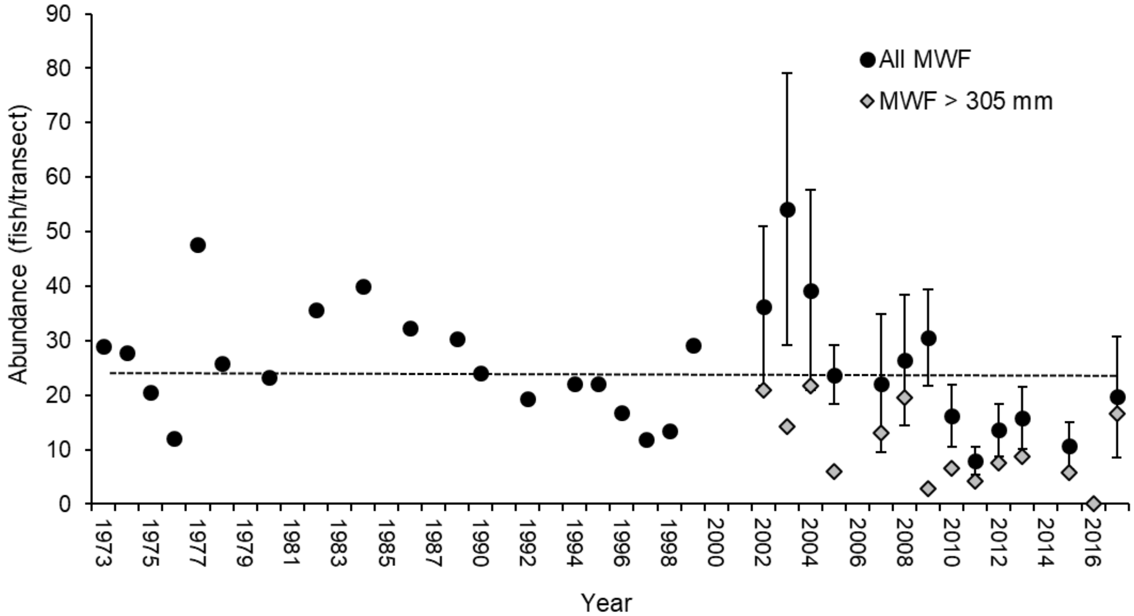

Figure 58. Mean density (all sizes and only fish > 305 mm) of Mountain Whitefish observed in 1-person snorkel transects in the main-stem Selway River, Idaho, from 1973 to 2017. Dashed line indicates mean density (24.3 fish/transect). Error bars represent 90% confidence intervals.

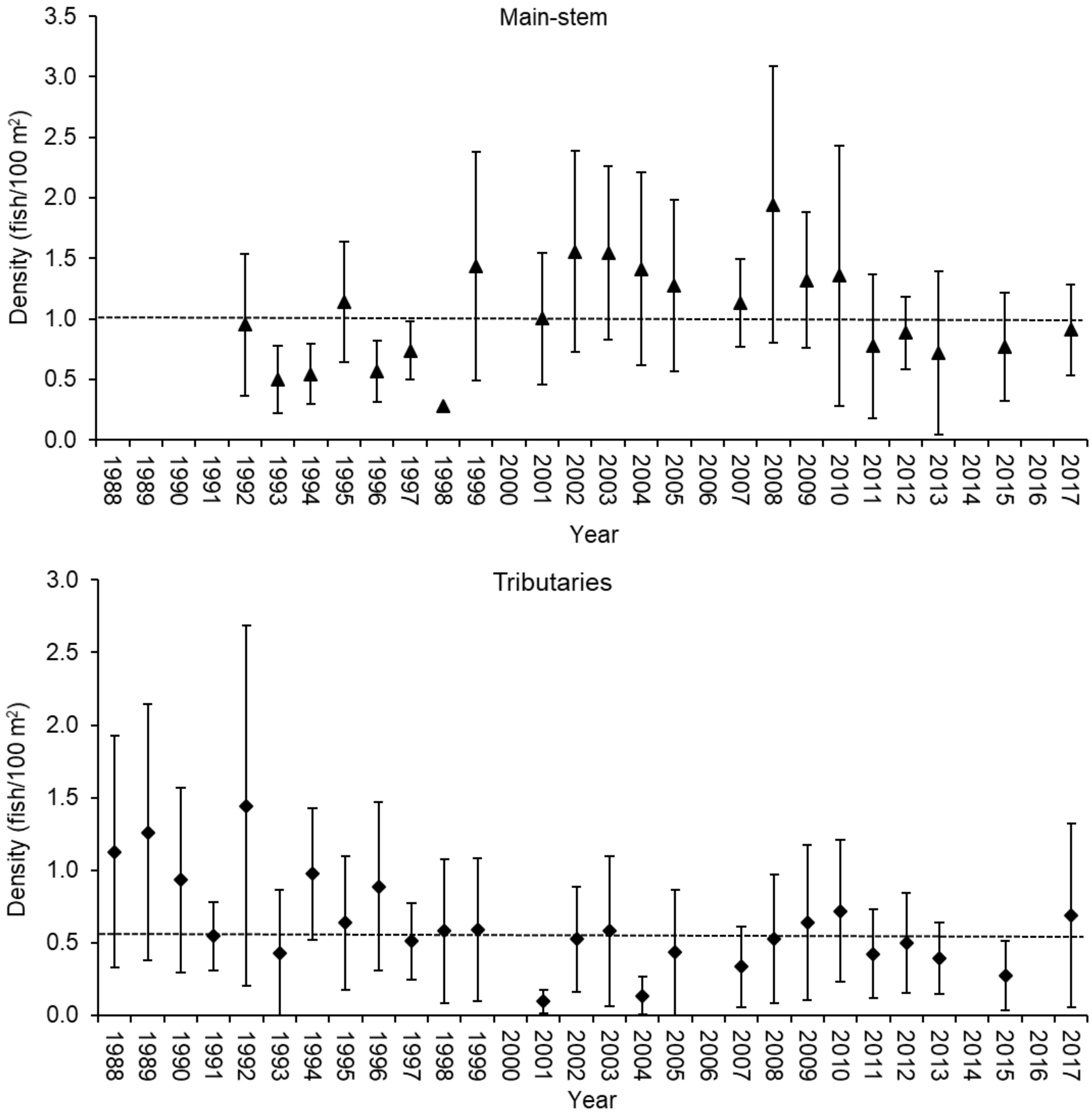

Figure 59. Mean density of Mountain Whitefish observed in General Parr Monitoring snorkel transects in the main-stem (1992 - 2017) and tributaries (1988 - 2017) of the main-stem Selway River, Idaho. Dashed lines indicate mean densities. Error bars represent 90% confidence intervals.

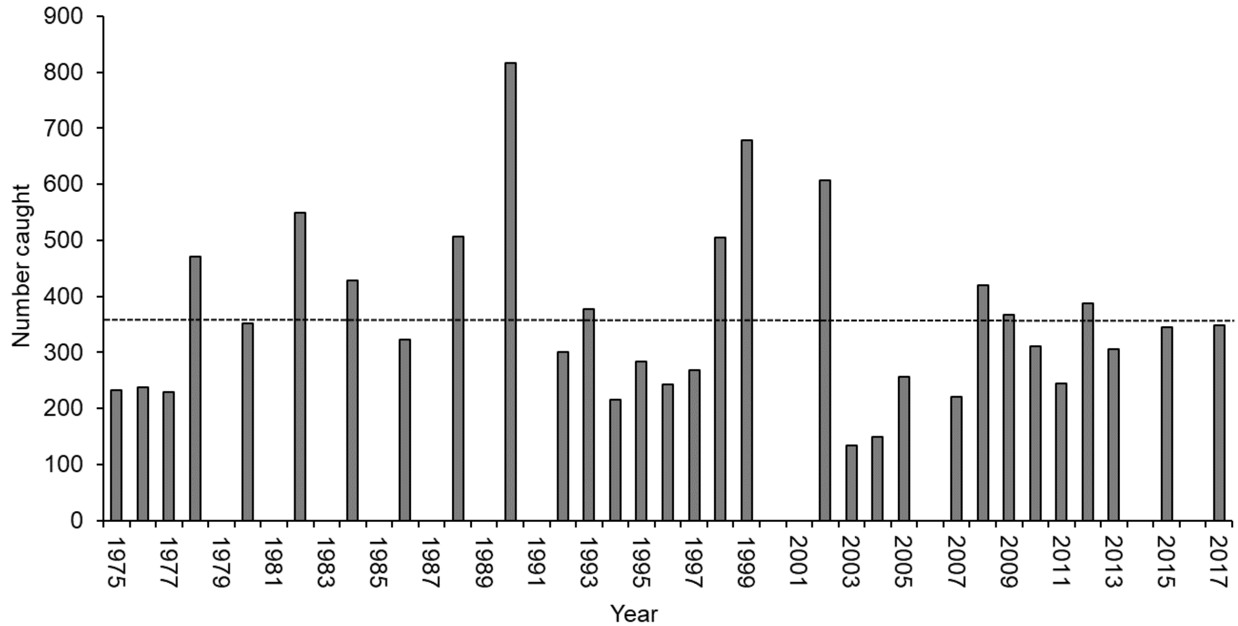

Figure 60. Number of Westslope Cutthroat Trout caught by hook-and-line surveys in the Selway River, Idaho, from 1975 to 2017.

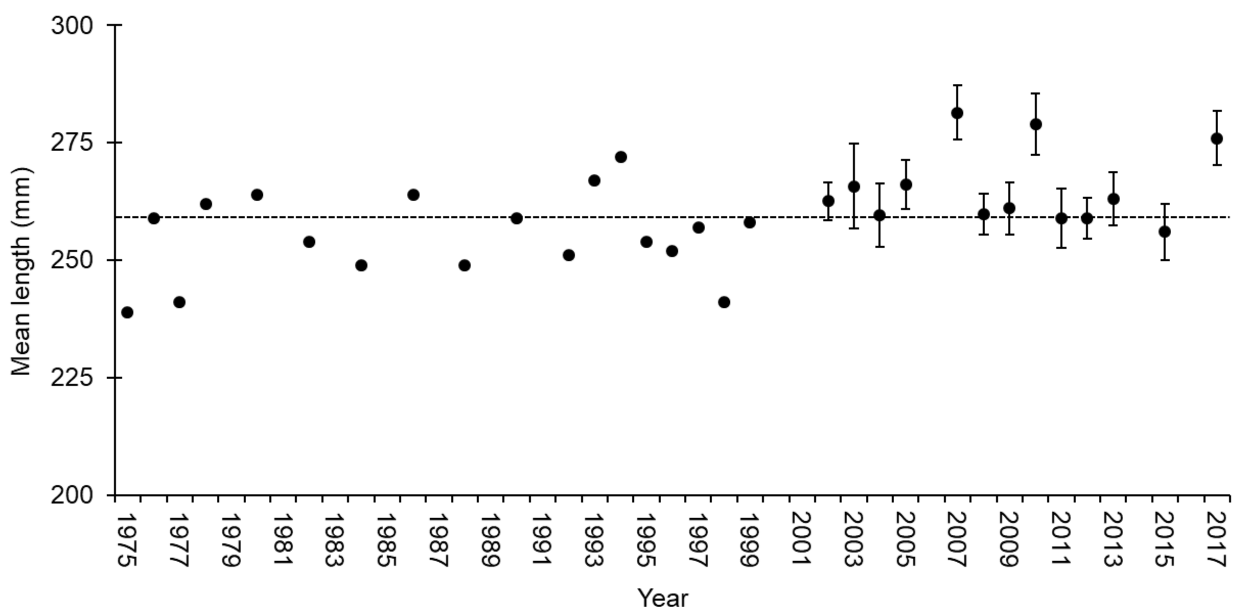

Figure 61. Mean total length of Westslope Cutthroat Trout caught by hook-and-line surveys in the Selway River, Idaho, from 1975 to 2017. Error bars represent 90% confidence intervals.

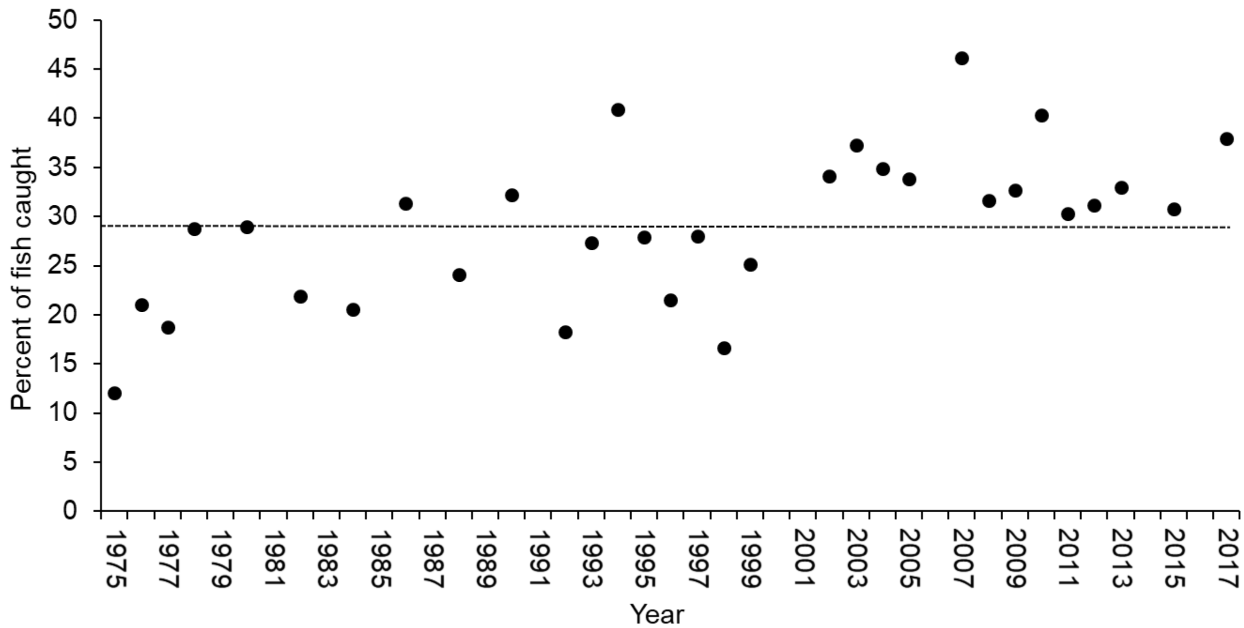

Figure 62. Percent of Westslope Cutthroat Trout > 305 mm caught by hook-and-line surveys in the Selway River, Idaho, from 1975 to 2017.

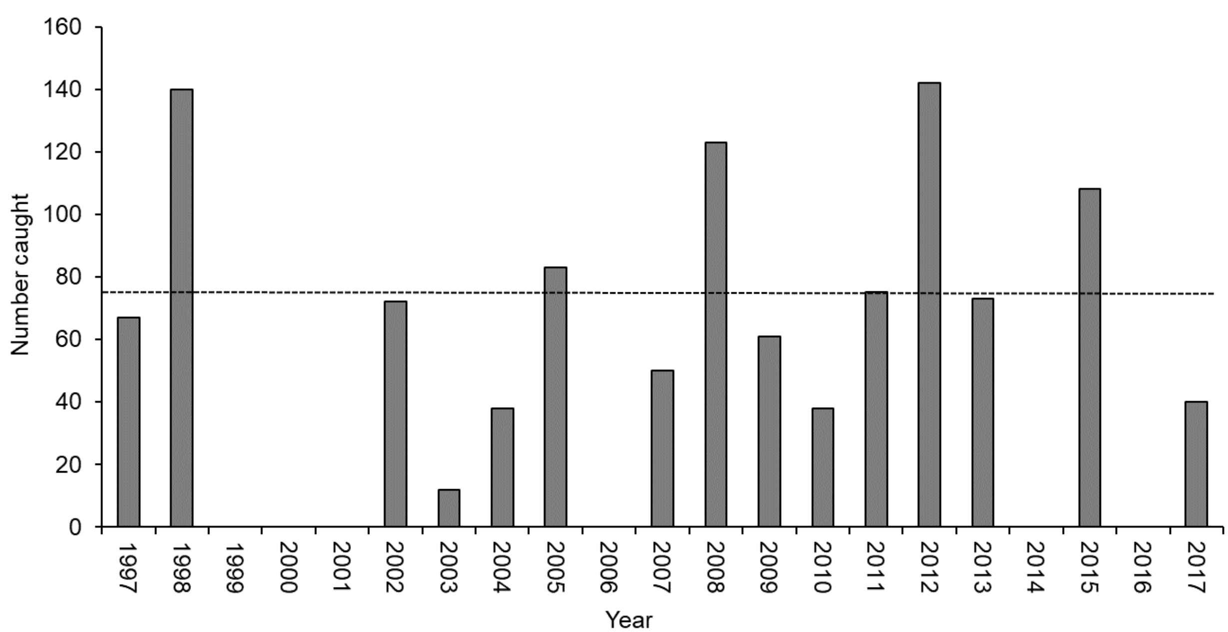

Figure 63. Number of Rainbow Trout caught by hook-and-line surveys in the Selway River, Idaho, from 1997 to 2017.

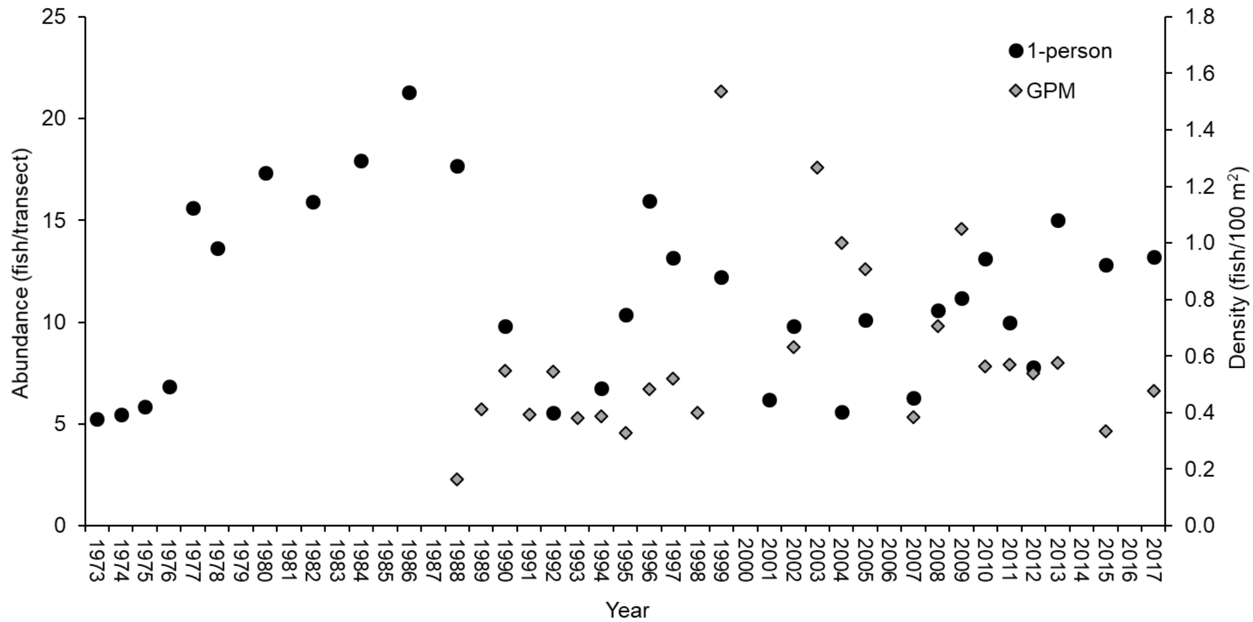

Figure 64. Comparison of Westslope Cutthroat Trout density (fish/transect) in 1-person transects and density (fish/100 m^2) in General Parr Monitoring transects on the main-stem Selway River, Idaho, from 1973 to 2017.

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EVALUATION OF FISH POPULATIONS IN THE SOUTH FORK CLEARWATER RIVER

ABSTRACT

Snorkel surveys were conducted on the main-stem South Fork Clearwater River in 2017 to assess trends in Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT), Rainbow Trout *O. mykiss* (RBT), and Mountain Whitefish *Prosopium williamsoni* (MWF) density and size distribution after implementing catch-and-release regulations for WCT in 2011. There has been a stable trend in mean WCT density over time (all fish and just those > 305 mm) for surveys conducted from 2000 to 2017. There has also been a stable trend in mean RBT density. For MWF, there was a significant declining trend in density of all size, but not for those individuals > 305 mm. One Smallmouth Bass *Micropterus dolomieu* was observed in the downstream-most transect. Westslope Cutthroat Trout density increased in other river systems after restrictive regulations were implemented. However, the lack of an increasing trend and overall low densities of WCT in the SFCR indicates that fishing mortality was not additive or they were a function of habitat and higher water temperatures found in this river system, rather than related to the regulation change. The majority of RBT observed in snorkel surveys were smaller than the steelhead smolts stocked into the SFCR. This suggests that the majority of the RBT observed in snorkel surveys were not hatchery smolts, and are primarily a combination of resident RBT and naturally produced juvenile steelhead. The declining trend in MWF density has been observed in the main-stem of other northern Idaho rivers, and in other rivers throughout the southern range of their distribution. While the direct cause is unknown, factors such as higher summer temperatures, prevalence of winter anchor ice, and disease (such as Proliferative Kidney Disease) may be impacting populations. With no significant trend in density of MWF > 305 mm, it appears the decline is primarily occurring in smaller sizes. If changes in habitat or temperature regimes are occurring, a decline in juvenile density may be an early indicator, and would explain declining trends in the overall population. The recent downward trend in MWF densities across northern Idaho rivers and other parts of their historic range warrants a more thorough evaluation. Smallmouth Bass were not observed in the main-stem SFCR sampling until 2014. Since then, they have been observed in low density with no evidence of reproduction. Based on our preliminary analysis, additional snorkel surveys are needed to more thoroughly evaluate long-term trends in density and density, and evaluate the 2011 regulation change.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT) are distributed throughout the South Fork Clearwater River (SFCR) drainage, occupying both the main-stem river and tributaries. Both resident and fluvial life history forms are present (Dobos 2015). While WCT are abundant in the other major Clearwater River tributaries (Lochsa, Selway, and North Fork Clearwater rivers), they are generally found in much lower densities in the SFCR (Cochnauer et al. 2001; Schill et al. 2005; Dobos 2015). This is mostly believed to be due to the higher water temperatures found in the SFCR system compared to the other tributaries (Dobos 2015). Fish populations in the SFCR have been regularly evaluated through snorkel surveys for a variety of projects. However, the main-stem river has only been surveyed three times since 2000. In 2011, daily bag limits on the main-stem SFCR were changed from two WCT (none < 356 mm) to a trout limit of six (all must have clipped adipose fin). This regulation change was implemented to protect WCT, and provide opportunity for anglers to keep hatchery Rainbow Trout *O. mykiss* (RBT) and residualized hatchery steelhead stocked in the SFCR basin. To evaluate impacts of this regulation change, and track long-term trends in density and distribution of SFCR resident fisheries, we have initiated a four-year sampling rotation to survey these main-stem transects on a two years on, two years off basis.

OBJECTIVES

1. Determine whether the density and size structure of Westslope Cutthroat Trout in the South Fork Clearwater River has improved since implementing catch-and-release regulations in 2011.

2. Evaluate trends in the density and size structure of Rainbow Trout and Mountain Whitefish *Prosopium williamsoni* (MWF) in the South Fork Clearwater River.

3. Evaluate whether the Smallmouth Bass *Micropterus dolomieu* (SMB) distribution is expanding in the South Fork Clearwater River.

STUDY AREA

Snorkel surveys were conducted on the main-stem SFCR, located in Idaho County, Idaho (Figure 65). The SFCR has a total drainage area of ~302,130 ha. Approximately 69% is located on National Forest lands, 23% is private ownership, and the remaining 8% is owned by other state and federal agencies, and the Nez Perce Tribe (Dobos 2014). Elevation of the main-stem SFCR ranges from 378 to 1,186 m. Mean discharge ranges from is 6 m^3/s in September to 90 m^3/s in May.

METHODS

Field sampling

A snorkel survey was conducted on the main-stem SFCR from July 30 to August 1, 2017. The river was divided into three sections: Lower - mouth to river kilometer (RKM) 30.0; Middle - RKM 30.0 to RKM 75.0; Upper - upstream of RKM 75.0. The delineations for these river sections were based on geomorphic differences as described by Dobos (2014). The Middle section consists of steep canyons and higher gradient compared to the Lower and Upper sections which were characterized as unconfined with large floodplains and lower gradients. A total of 21 snorkel transects were surveyed (Figure 65; Appendix A). To maintain consistency, we surveyed the same transects that were snorkeled from 2000 to 2014 (Hand et al. 2013). These are a subset of transects surveyed as part of the Idaho Supplementation Studies program, developed to evaluate supplementation as a potential tool for recovery of Snake River basin Chinook Salmon *O. tshawytscha* (Lutch et al. 2003). Transects were surveyed using the standard snorkeling methodologies outlined in Apperson et al. (2015). This technique entails snorkeling downstream using an appropriate number of snorkelers to cover the entire width of the river to allow for the calculation of fish densities. All fish observed were counted, and length was estimated to the nearest inch for all game species. Non-game species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson, 50-m, reel-style, fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. The distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead and resident RBT are indistinguishable and are collectively referred to as “RBT”. This report focuses on WCT, RBT, and MWF. Results and analysis of data collected on other species in 2017 can be found in Putnam et al. (2018).

Data analysis

We used least squares regress to evaluate trends in density (fish/100 m^2) of WCT, RBT, and MWF of all sizes. We also assessed trends in size structure by evaluating density of WCT and MWF > 305 mm using least squares regression, and for RBT by developing a length-frequency distribution. All regression analysis used survey year (2000 - 2017) as the independent variable and $\log_e$ transformed density as dependent variables (Maxell 1999; Kennedy and Meyer 2015). The intrinsic rate of change in the population (r_{intr}) was determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance, where the trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$. Distributions of WCT, RBT, and MWF were visually represented by plotting mean density for each transect on maps of the survey area using GIS software.

<u>RESULTS</u>

Chinook Salmon (n = 348), RBT (n = 348), WCT (n = 9), Brook Trout *Salvelinus fontinalis* (BKT; n = 2), MWF (n = 82), and SMB (n = 2) were observed during the snorkel survey conducted in 2017 (Table 24). Water temperatures averaged 20.4 °C.

Westslope Cutthroat Trout

Westslope Cutthroat Trout were observed in four transects in 2017 (Figure 66). No more than four WCT were observed in any transect. Mean density (0.02 fish/100 m^2) was the lowest observed of the four surveys conducted since 2000 (Table 25); however, there was a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in density since 2000 (Table 26). Mean density of WCT > 305 mm (0.004/100 m^2) was four times higher than 2014; however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in density of WCT > 305 mm since 2000 (Table 26). Additionally, mean length of WCT observed was lower in 2017 than 2010 and 2014 (Table 27).

Rainbow Trout

Rainbow Trout were observed in 17 of the 21 transects snorkeled in 2017, with the highest densities occurring between Johns Creek and Newsome Creek (Table 28 and Figure 67). Mean RBT density was lower than the long-term average density for surveys conducted since 2000 (Table 28); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in RBT density since 2000 (Table 26).

Mean RBT length was ~98 mm. Fifty-six percent of RBT observed were < 102 mm in length, the minimum length of hatchery steelhead smolts stocked into the SFCR in 2017 (Figure 68). One percent of RBT observed were larger than the steelhead smolts stocked into the SFCR in 2017.

Mountain Whitefish

Mountain Whitefish were distributed through most of the SFCR, except the downstream-most transects (Figure 69). The mean density of MWF was the lowest of the four surveys conducted since 2000 (Table 29), and there was a statistically significant declining trend ($r_{intr} < 0$) in density (Table 26). Mean density of MWF > 305 mm (0.087/100 m^2) was three times lower than 2014, and was the lowest of any survey conducted since 2010. However there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in density of MWF > 305 mm (Table 26).

Smallmouth Bass

One SMB was observed in transect (18.2KM) in 2017 (Table 24). The only other time a SMB was observed in the SFCR was in the downstream-most transect (3.5KM) in 2014.

DISCUSSION

Westslope Cutthroat Trout

The observed WCT mean density in 2017 was the lowest of any survey we have conducted and an increasing trend in density has not occurred since catch-and-release limits were implemented. This contrasts with the increasing trends observed in the main-stem Selway, Lochsa, St. Joe, and Coeur d'Alene rivers after regulation changes were implemented in those rivers (*See* Selway River and Lochsa River chapters in this report; Ryan et al. 2020). In all of these rivers, changes in density were observed within a year or two after catch-and-release limits were implemented. This suggests that factors others than harvest are limiting WCT density in the SFCR. Based on the temperatures observed in this survey, it is likely temperature plays a large role in the low densities of WCTin the SFCR. The average temperature in the SFCR was 2 - 4°C warmer than the Lochsa, Selway, and St. Joe rivers in 2017 (See Selway River and Lochsa River chapters in this report; Ryan et al. 2020). Additionally, temperatures were detected that exceeded thermal tolerances (22°C) for adult WCT (Bjornn and Reiser 1991; Bear et al. 2007; Dobos 2015). Westslope Cutthroat Trout in the SFCR migratelong distances to seek out cooler temperatures in tributaries and upper reaches of the SFCR during the time of our survey (Dobos 2015). However, we did not see many WCT in the Upper section of the SFCR. Previous studies have suggested that this section contains poorer quality habitat compared to the Middle section, which could account for lower densities (Dobos 2014). Additional surveys will allow for a more thorough analysis of population trends and potential impacts of the regulation change.

There was no significant trend in mean density of WCT > 305 mm from 2000 to 2017. The regulation change does not appear to have impacted the size structure of WCT in the main-stem SFCR.

Rainbow Trout

The mean density of RBT observed in the SFCR has fluctuated between survey years resulting in no statistical trend in density since 2000. However, we caution that this analysis is based on four years of data. Rainbow Trout density in the SFCR was 5 and 10 times higher than the main-stem Selway River (0.13/100 m^2) and Lochsa River (0.25/100 m^2; *See* Selway River and Lochsa River chapters in this report). The higher density in the SFCR could be explained by the annual stocking of ~1.2 million hatchery steelhead smolts in the SRCR basin. However, the majority of RBT observed in SFCR snorkel surveys were smaller than the steelhead smolts stocked into the SFCR. The mean length of RBT observed in the SFCR was similar to the Selway River (~101 mm), but higher than the Lochsa River (~71 mm), both of which are wild populations. Additionally, few RBT with clipped adipose fins were observed during snorkel surveys, although this mark is often difficult to see, especially during downstream surveys (Apperson et al. 2015). This suggests that the majority of the RBT observed in snorkel surveys were not hatchery smolts, and are primarily a combination of resident RBT and naturally produced juvenile steelhead.

The regulation change does not appear to have impacted the size structure of RBT in the main-stem SFCR. We also recommend conducting additional snorkel surveys in areas were higher densities of RBT are observed to evaluate what percent have adipose fin clips. This will provide insight into what extent hatchery steelhead contribute to the resident fishery.

Mountain Whitefish

A declining trend was observed in the mean density of MWF in the main-stem SFCR for surveys conducted since 2000. We must caution that this analysis is based on four years of data, and confined to the main-stem SFCR. Declines in MWF have been observed in other northern Idaho rivers, including the Selway River and Lochsa River (*See* Selway River and Lochsa River chapters in this report). Recently, declines in MWF have been documented in other populations across the southern portion of their range as well, including the Big Lost River and Kootenai River, Idaho, the Yampa River, Colorado, and the Madison River, Montana (Paragamian 2002; IDFG 2007; Boyer 2016). While the direct cause of these declines has not been identified, these declines have been linked to occurrences of low flows and higher water temperatures (Brinkman et al. 2013). These studies also suggested that habitat alteration, irrigation, nonnative fish interactions, disease, and harvest are also likely contributing to declines in MWF populations (IDFG 2007; Boyer 2016). While the SFCR has a long history of habitat alteration and agricultural use which may have had historic impacts on local fish populations, these factors probably do not account for the recent declines.

More recently, environmental factors such as higher summer water temperatures and severe winter conditions are more likely contributing to the declining trends in MWF density in the SFCR. Increased water temperatures could affect MWF populations in several ways. Warmer temperatures may impact populations by moving more fish out of the main-stem and larger tributaries (where we surveyed) and/or through increased mortality (Hunt 1992; Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Mean monthly summer air temperatures have been above normal every year except one since 1996 (NOAA 2021). Additionally, severe outbreaks of Proliferative Kidney Disease (PKD) have been observed in Montana, and the disease is known to be present in Idaho (Phillips 2016; Hutchins et al. 2021). While no major fish kills have been directly observed, minor die-offs have been observed in in Idaho rivers during summer months. Thus, PKD could impact populations through lower level mortality. As such, increased water temperatures could impact MWF movement, survival, and recruitment before some other species.

In addition to warmer summer temperatures, climate change may actually cause more severe winter conditions through the increase in prevalence of anchor ice by reducing snow cover, which insulates against anchor ice formation (Butler 1979). Specifically, anchor ice which forms on the bottom of river beds can directly affect fish populations through direct mortality (increased stress, stranding, etc.) as well as impacts to redds and benthic invertebrate communities that serve as food sources (Butler 1979; Jakober et al. 1998; Brown et al. 2011). Anchor ice is known to be prevalent in SFCR tributaries (especially Red River), and lower snowfall could increase its formation and duration, and therefore its potential impacts on fish and habitat.

In contrast to overall density, there was no significant trend in density of MWF > 305 mm. This suggests that the trends in MWF populations observed in the SFCR are size dependent, with declines occurring at smaller sizes. Within the SFCR, fewer juvenile MWF have been observed during snorkel surveys over the last 10 years compared to historic surveys (*IDFG unpublished data*; Putnam et.al 2017*; Scott Putnam, personal communication*). Fish populations are often limited by recruitment, and changes in juvenile survival would have long-lasting effects on the population (Bradford and Cabana 1997; Pope et al. 2010). If changes in habitat or temperature regimes are occurring, a decline in juvenile density may be an early indicator, and would explain why we are seeing declining trends in the overall population.

Additional surveys will allow for a more thorough analysis of trends in MWF populations in the main-stem SFCR. However, the apparent downward trend in MWF density across the Clearwater River drainage and other parts of their historic range warrants a more detailed analysis.

Smallmouth Bass

Smallmouth Bass were not observed in the main-stem SFCR sampling until 2014. Since then, they have been observed in low density with no evidence of reproduction. They have also been recently observed in the lower reaches of the Lochsa River and North Fork Clearwater River (*See* Lochsa River chapter in this report; Hand et al. 2020). Smallmouth Bass colonization of salmonid spawning and rearing habitat has been documented throughout the Columbia River Basin (Lawrence et al. 2014; Rubenson and Olden 2017). Potential increases in SMB distribution is of concern, as these non-native fish are a substantial predator of juvenile anadromous fish and can impact resident fish populations as well (Tabor et al. 1993; Naughton et al. 2004). Future surveys in the SFCR should continue to record observations of SMB, as they may experience a climate change related spread throughout the Clearwater River system (Rahel and Olden 2008).

MANAGEMENT RECOMMENDATIONS

1. Continue to evaluate trends in density and the size structure of Westslope Cutthroat Trout in the South Fork Clearwater River on a two year on, two year off basis.

2. Continue to evaluate trends in Rainbow Trout and Mountain Whitefish size and density in the South Fork Clearwater River.

3. Make attempts to assess the proportion of adipose clipped RBT during snorkel surveys in select areas.

4. Continue to monitor Smallmouth Bass distribution and density in the South Fork Clearwater River to assess whether upstream colonization is increasing.

Table 24. Fish density (fish/100 m^2) by transect, for snorkel surveys conducted on the main-stem South Fork Clearwater River, Idaho, in 2017.

				Density						
River section	Transect name	Temp °C	Visibility (m)	RBT	Chinook Salmon	Westslope Cutthroat Trout	Mountain Whitefish	Bull Trout	Brook Trout	Smallmouth Bass
Lower	3.5 KM	24.0	1.1	0.05	0.11	0.00	0.00	0.00	0.00	0.00
	8.5 KM	24.0	1.3	0.00	0.17	0.00	0.00	0.00	0.00	0.00
	13.4 KM	25.0	1.1	0.24	0.59	0.00	0.00	0.00	0.00	0.00
	18.2 KM	25.0	1.8	0.00	0.29	0.04	0.31	0.00	0.00	0.06
	23.0 KM	24.0	2.6	0.00	1.09	0.00	0.31	0.00	0.00	0.00
	28.5 KM	23.0	1.1	0.48	0.00	0.00	0.00	0.00	0.00	0.00
Middle	33.7 KM	23.0	1.4	1.45	1.81	0.00	0.14	0.00	0.00	0.00
	38.5 KM	22.0	1.6	3.87	0.93	0.00	0.27	0.00	0.13	0.00
	43.9 KM	18.0	1.3	0.33	0.12	0.00	0.33	0.00	0.00	0.00
	48.7 KM	19.0	1.1	0.85	1.33	0.14	0.85	0.00	0.05	0.00
	53.0 KM	18.0	1.2	0.35	0.04	0.00	0.35	0.00	0.00	0.00
	58.2 KM	19.5	1.4	3.20	0.50	0.06	0.19	0.00	0.00	0.00
	63.7 KM	19.0	2.7	5.17	4.90	0.00	0.16	0.00	0.00	0.00
	68.6 KM	19.0	2.3	1.59	0.79	0.00	0.40	0.00	0.00	0.00
	73.7 KM	21.0	2.5	2.66	0.00	0.00	0.00	0.00	0.00	0.00
Upper	78.3 KM	---	1.0	2.69	2.69	0.00	0.10	0.00	0.00	0.00
	83.9 KM	19.0	1.8	2.47	1.76	0.22	0.44	0.00	0.00	0.00
	88.7 KM	18.0	1.3	0.34	1.68	0.00	0.00	0.00	0.00	0.00
	93.9 KM	18.0	1.7	0.36	0.36	0.00	0.12	0.00	0.00	0.00
	98.7 KM	15.0	1.3	0.00	0.26	0.00	0.00	0.00	0.00	0.00
	103.2 KM	15.0	1.9	0.28	2.50	0.00	0.37	0.00	0.00	0.00
	Mean			1.26	1.04	0.02	0.21	0.00	0.01	0.00
	90% CI			0.54	0.43	0.02	0.08	---	0.01	0.00

Table 25. Density (fish/100 m^2) of Westslope Cutthroat Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2017.

River section	Transect	2000	2010	2014	2017
Lower	3.5 KM	0.00	0.00	0.00	0.00
	8.5 KM	0.00	0.00	0.00	0.00
	13.4 KM	0.00	0.00	0.00	0.00
	18.2 KM	0.07	0.00	0.00	0.04
	23.0 KM	0.00	0.00	0.00	0.00
	28.5 KM	0.00	0.00	0.00	0.00
Middle	33.7 KM	0.00	0.00	0.06	0.00
	38.5 KM	0.44	0.00	0.00	0.00
	43.9 KM	0.00	0.00	0.00	0.00
	48.7 KM	0.05	0.07	0.00	0.14
	53.0 KM	0.00	0.08	0.00	0.00
	58.2 KM	0.14	0.15	0.23	0.06
	63.7 KM	0.00	0.41	0.29	0.00
	68.6 KM	0.00	0.00	0.00	0.00
	73.7 KM	0.00	0.33	0.16	0.00
Upper	78.3 KM	0.10	0.12	0.00	0.00
	83.9 KM	0.08	0.48	0.10	0.22
	88.7 KM	0.00	0.63	0.00	0.00
	93.9 KM	0.00	0.18	0.12	0.00
	98.7 KM	0.13	0.32	0.00	0.00
	103.2 KM	0.04	0.14	0.00	0.00
	Mean	0.05	0.14	0.05	0.02
	90% CI	0.04	0.07	0.03	0.02

Table 26. Intrinsic rate of change (r_{intr}) in density (fish/100 m^2) for Westslope Cutthroat Trout, Rainbow Trout, and Mountain Whitefish for snorkel surveys conducted in the South Fork Clearwater River, Idaho, from 2000 to 2017. Significance was set at α = 0.10.

Species	r_{intr} estimate	90% CI lower	90% CI upper
Westslope Cutthroat Trout			
all sizes	-0.037	-0.233	0.159
> 305 mm	0.339	-0.245	0.923
Rainbow Trout	-0.091	-0.283	0.101
Mountain Whitefish			
all sizes	-0.064	-0.116	-0.012
> 305 mm	-0.071	-0.175	0.033

Table 27. Minimum, maximum, and mean length of Westslope Cutthroat Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2017.

		Length (mm)		
Year	*n*	Min	Max	Mean
2000	35	76	254	135
2010	49	76	432	214
2014	8	127	305	270
2017	9	76	406	186

Table 28. Density (fish/100 m^2) of Rainbow Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2017.

River section	Transect	2000	2010	2014	2017
Lower	3.5 KM	0.00	0.00	0.58	0.05
	8.5 KM	0.00	0.00	0.00	0.00
	13.4 KM	0.00	0.07	0.00	0.24
	18.2 KM	1.34	0.00	0.00	0.00
	23.0 KM	0.13	0.17	0.24	0.00
	28.5 KM	1.57	0.00	0.10	0.48
Middle	33.7 KM	0.00	0.04	6.68	1.45
	38.5 KM	8.32	3.38	5.33	3.87
	43.9 KM	0.74	0.26	1.07	0.33
	48.7 KM	5.09	0.34	0.54	0.85
	53.0 KM	1.52	1.07	0.63	0.35
	58.2 KM	3.54	0.45	5.05	3.20
	63.7 KM	5.19	1.56	5.17	5.17
	68.6 KM	8.43	0.63	6.41	1.59
	73.7 KM	6.14	0.99	5.02	2.66
Upper	78.3 KM	8.09	1.22	5.44	2.69
	83.9 KM	2.57	0.96	0.70	2.47
	88.7 KM	3.32	1.96	0.38	0.34
	93.9 KM	2.13	1.14	0.00	0.36
	98.7 KM	3.31	1.34	0.00	0.00
	103.2 KM	2.02	2.03	0.00	0.28
	Mean	3.02	0.84	2.06	1.26
	90% CI	1.02	0.32	0.93	0.54

Table 29. Density (fish/100 m^2) of Mountain Whitefish observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2017.

River section	Transect	2000	2010	2014	2017
Lower	3.5 KM	0.00	0.00	0.58	0.00
	8.5 KM	0.09	0.07	0.00	0.00
	13.4 KM	0.19	0.00	0.11	0.00
	18.2 KM	0.61	0.17	1.82	0.31
	23.0 KM	2.90	0.00	1.04	0.31
	28.5 KM	1.80	0.06	0.68	0.00
Middle	33.7 KM	0.40	0.26	0.06	0.14
	38.5 KM	0.81	4.17	0.27	0.27
	43.9 KM	1.04	0.09	0.45	0.33
	48.7 KM	1.95	0.27	1.21	0.85
	53.0 KM	1.58	0.15	1.26	0.35
	58.2 KM	1.07	0.25	0.00	0.19
	63.7 KM	4.60	0.14	0.44	0.16
	68.6 KM	0.70	0.21	1.28	0.40
	73.7 KM	2.52	0.00	1.88	0.00
Upper	78.3 KM	1.55	0.49	0.45	0.10
	83.9 KM	1.41	0.17	0.91	0.44
	88.7 KM	1.59	0.25	0.96	0.00
	93.9 KM	1.68	0.42	0.12	0.12
	98.7 KM	1.03	0.24	0.00	0.00
	103.2 KM	1.62	2.17	0.09	0.37
	Mean all sizes	1.38	0.46	0.65	0.21
	90% CI	0.38	0.35	0.21	0.08
	Mean > 305 mm	0.39	0.15	0.24	0.09
	90% CI	0.27	0.10	0.20	0.05

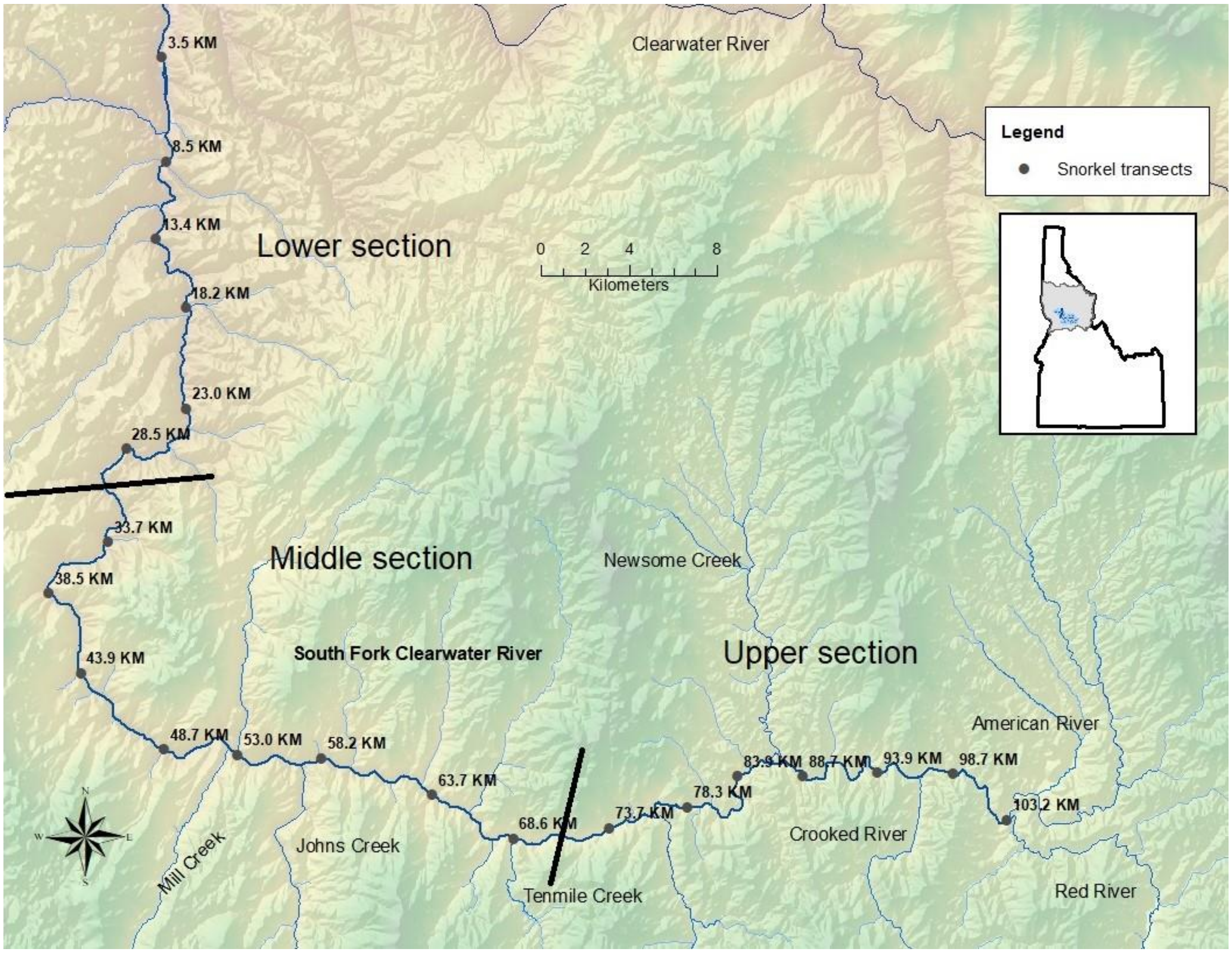

Figure 65. Location of snorkel transects surveyed on the South Fork Clearwater River, Idaho, in 2017.

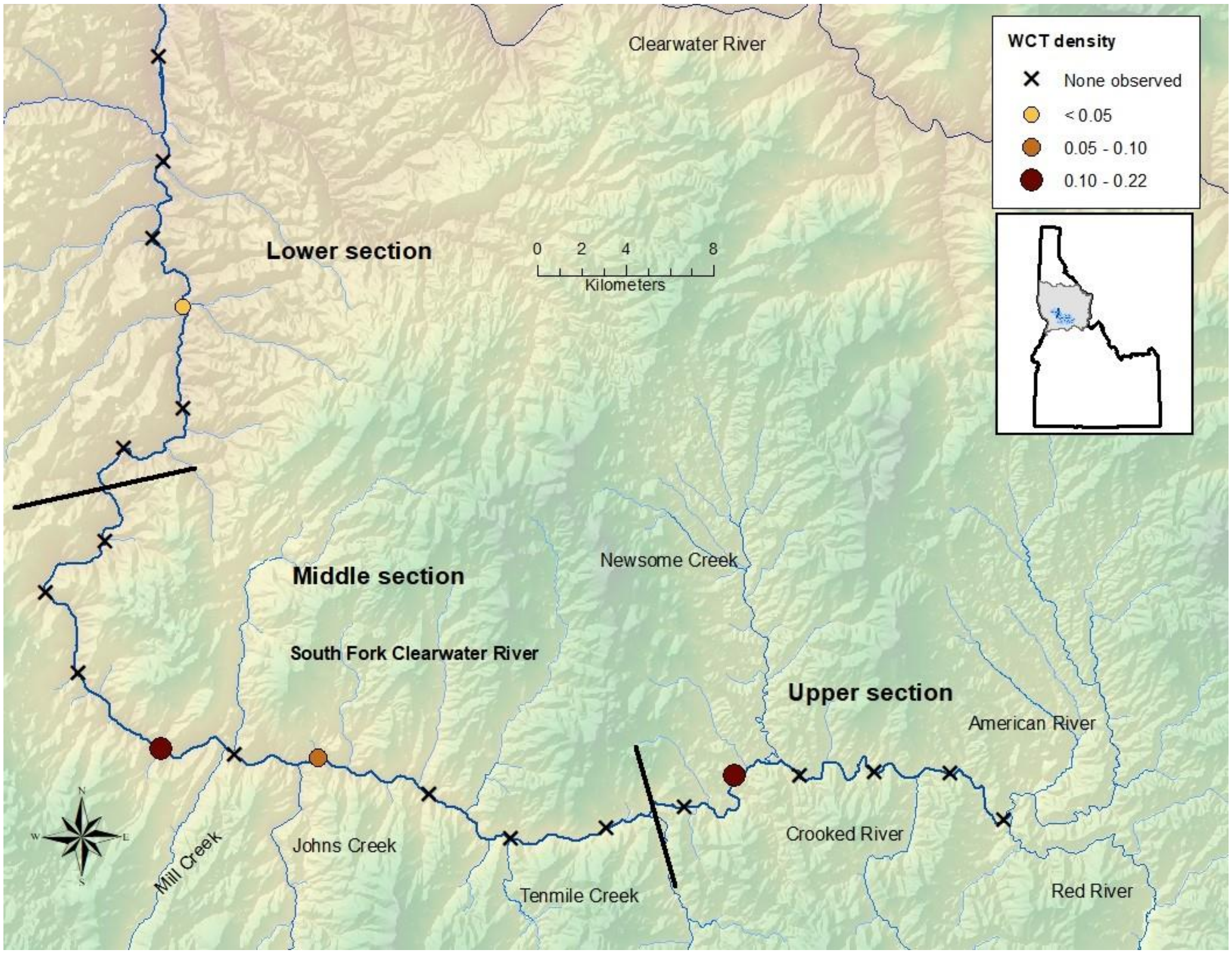

Figure 66. Density (fish/100 m^2) of Westslope Cutthroat Trout (WCT) in each snorkel transect surveyed in the South Fork Clearwater River, Idaho, in 2017.

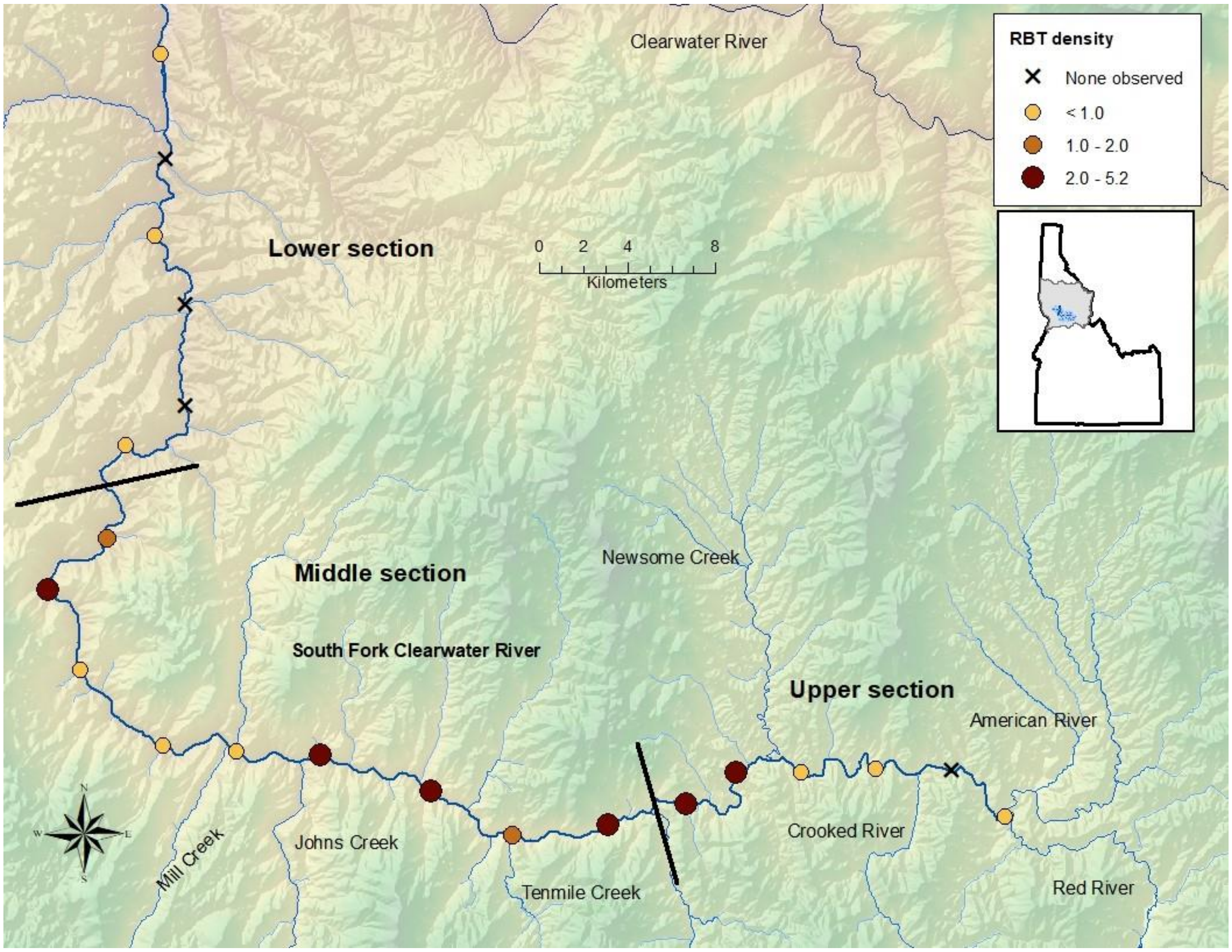

Figure 67. Density (fish/100 m^2) of Rainbow Trout in each snorkel transect surveyed in the South Fork Clearwater River, Idaho, in 2017.

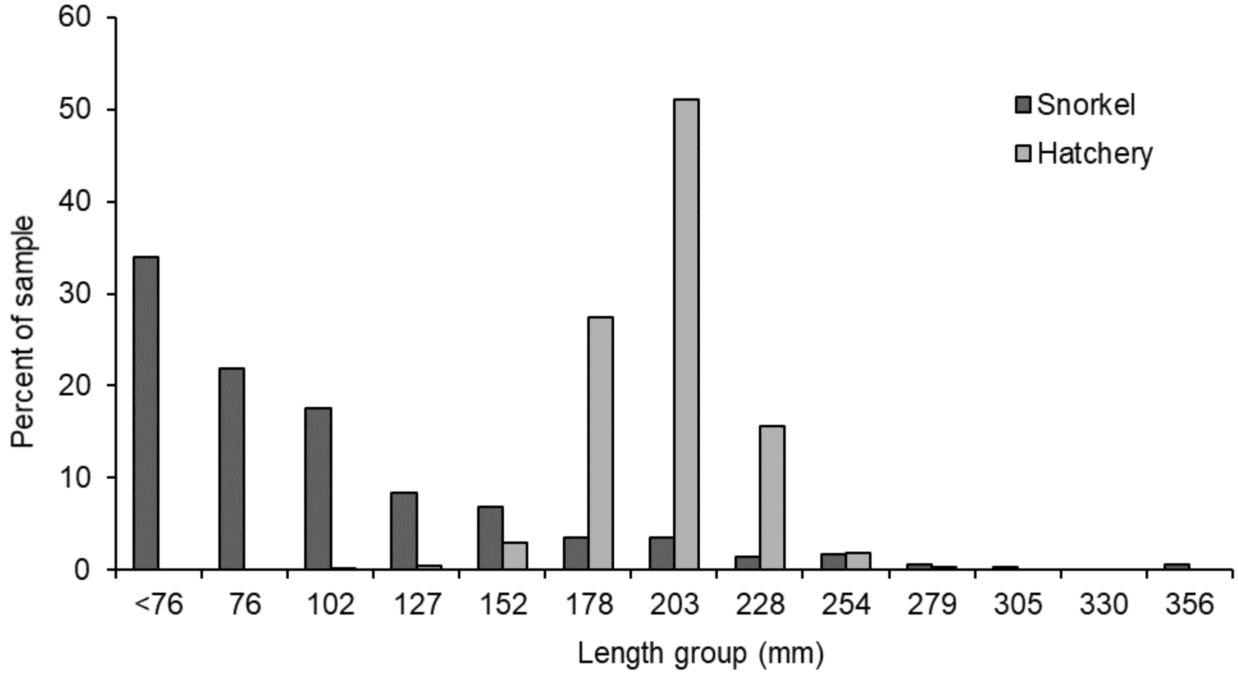

Figure 68. Length-frequency distributions of Rainbow Trout observed in snorkel transects and steelhead smolts stocked in the South Fork Clearwater River, Idaho, in 2017.

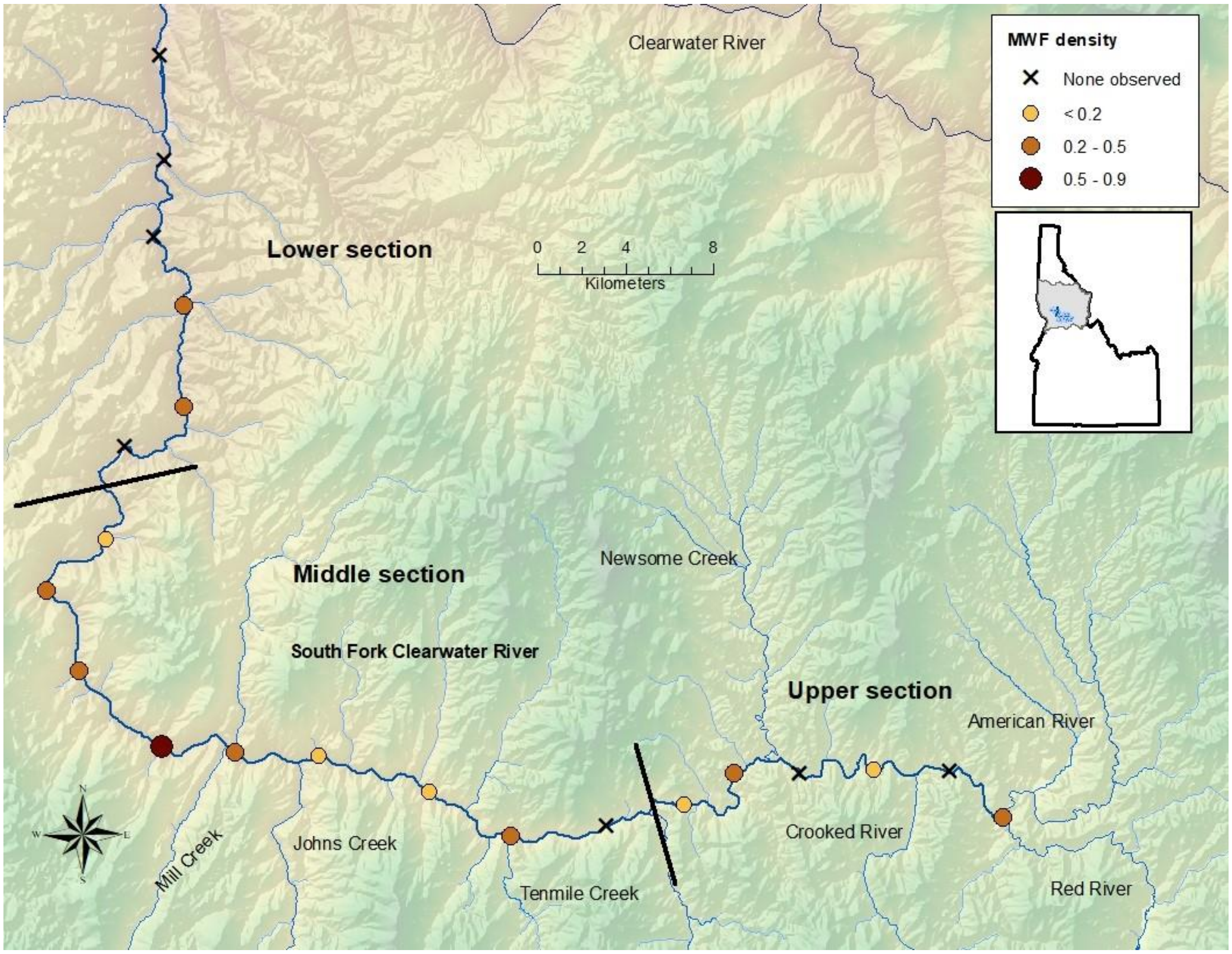

Figure 69. Density (fish/100 m^2) of Mountain Whitefish (MWF) in each snorkel transect surveyed in the South Fork Clearwater River, Idaho, in 2017.

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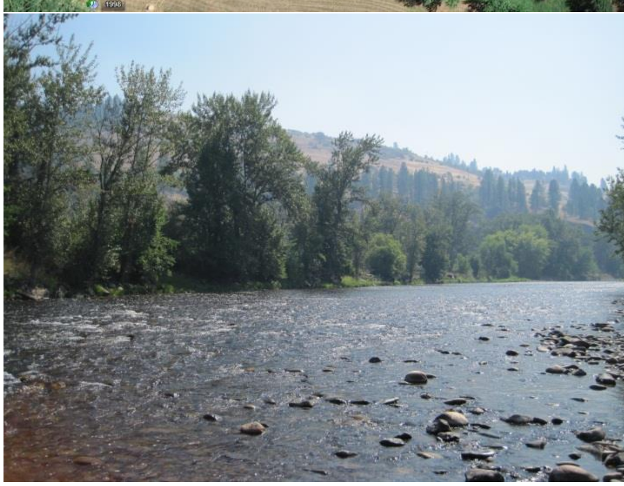
Appendix A. SFCW 3.8 site is best accessed by Chandler Lane which is 2.4 road miles upstream from Kooskia on Highway 13. There will be a parking area on the left soon after you pull off the highway. The snorkel site is a riffle downstream of the parking area. The best way to access the site is to cross the river near the parking area and walk down the bank to the site. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	46.112643	-115.982596	43m
Lower End:	46.112954	-115.982976	

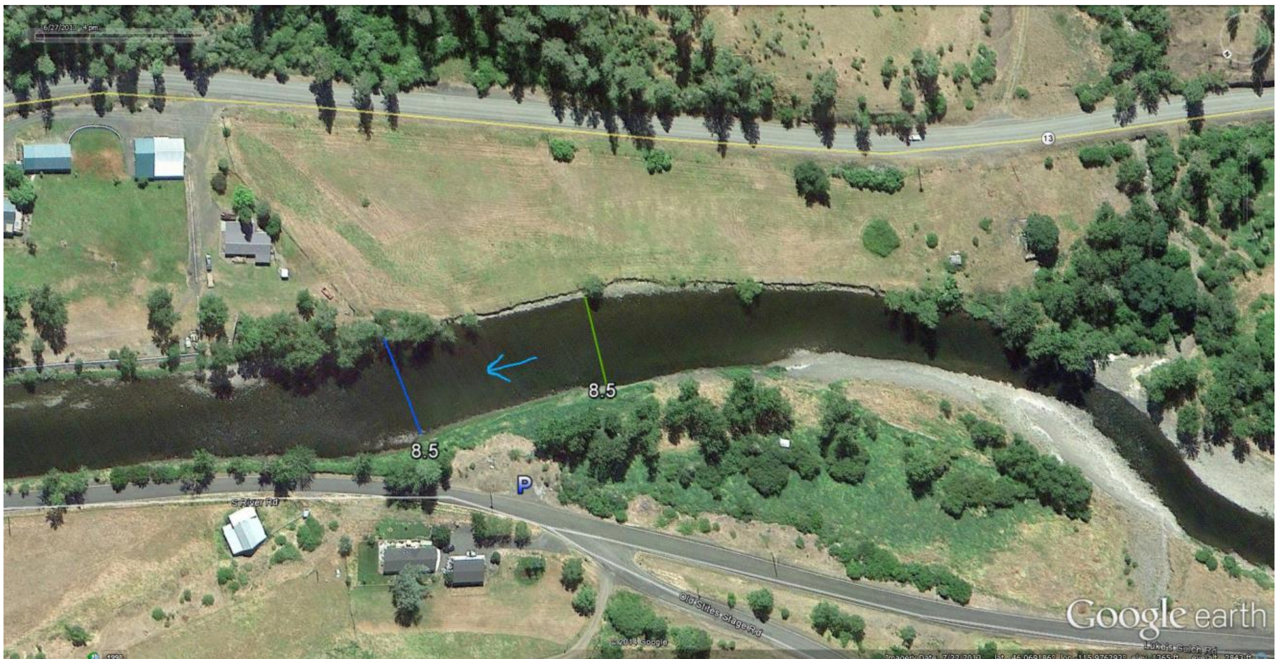


Bottom of Site Looking Up

Top of Site Looking Down

Appendix B. SFCW 8.5 site is best accessed by crossing the river at Stites (3.8 road miles upstream of Kooskia on Highway 13). In Stites, take a right on to Bridge St to cross the river. After crossing the river take the first left onto N River Road; drive 0.7 miles and veer left at the fork onto S River Road. Drive one mile to where the Old Stites Stage Road forks and there will be a pullout next to the river. The site is a run with a riffle on upstream and downstream sides next to the pullout. Site was snorkeled downstream in 2014.

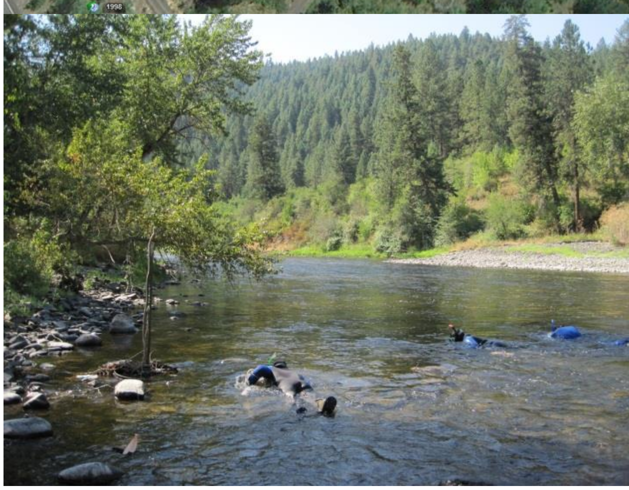
WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	46.069126	-115.976607	77m
Lower End:	46.069756	-115.976232	



Bottom of Site Looking Up Top of Site Looking Down

Appendix C. SFCW 13.4 site is located approximately 8.3 road miles upstream from Kooskia on Highway 13. There is a pullout next to the river directly upstream from a couple houses. Walk down stream to the snorkel site being mindful of private property. The site is a run with riffles on both upstream and downstream ends. Site was snorkeled upstream in 2014.

WGS84	Latitude	Longitude	Length of Site
Upper End:	46.036904	-115.982139	74m
Lower End:	46.037642	-115.98213	




Bottom of Site Looking Up Top of Site Looking Down

Appendix D. SFCW 18.2 site is located approximately 11.0 road miles upstream of Kooskia on Highway 13. There is a small pullout for parking directly after Sally Ann Creek Road on the side of the highway opposite of the river. Cross the highway upstream of private property and down to the river. The snorkel site is a pool that starts at the mouth of Sally Ann Creek. Site was snorkeled downstream in 2014.

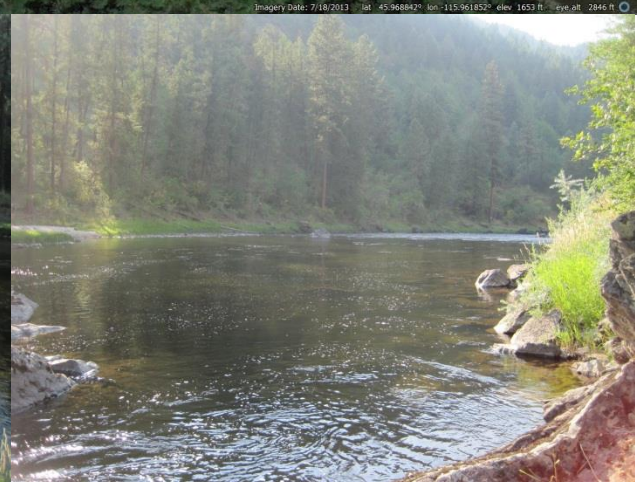
WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	46.009943	-115.96451	107m
Lower End:	46.009927	-115.964540	

Bottom of Site Looking Up

Top of Site Looking Down

Appendix E. SFCW 23.0 parking for site is approximately 14.3 road miles upstream of Kooskia on Highway 13 (0.8 miles upstream of Sears Creek Road.) There is a pullout on the side of the highway adjacent to the river. The site is a pool approximately 160 meters downstream of the pullout. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.968297	-115.961863	62m
Lower End:	45.968725	-115.962427	

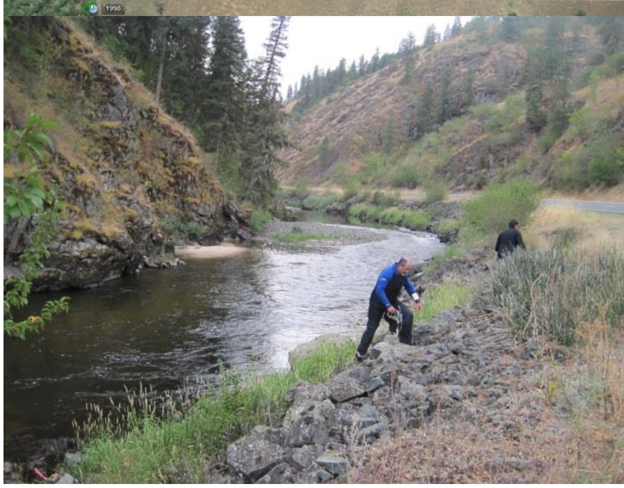


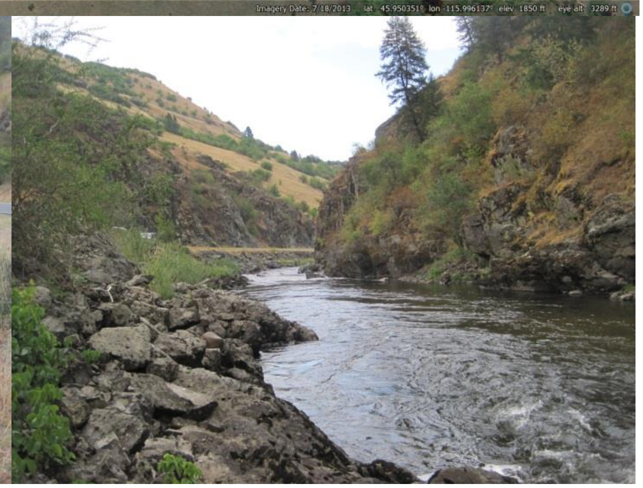
Bottom of Site Looking Up

Top of Site Looking Down

Appendix F. SFCW 28.5 site is located approximately 17.2 road miles upstream from Kooskia (2.1 miles upstream from Highway 13/14 Junction) on Highway 14. There is a pullout on the side of the highway adjacent to the river. The site is a fast pool next to the pullout. The site starts at a fast riffle and ends at a small rapid; **take caution crossing the riffle and be sure to get to shore before the rapid**. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.950876	-115.996805	55m
Lower End:	45.951102	-115.996219	





Bottom of Site Looking Up

Top of Site Looking Down

Appendix G. SFCW 33.7 site is located approximately 20.7 road miles upstream from Kooskia on Highway 14. There is a private bridge crossing the river just downstream of the site. Directly upstream of the bridge is a dirt road that goes down to the river. The site is a riffle near the end of the dirt road.

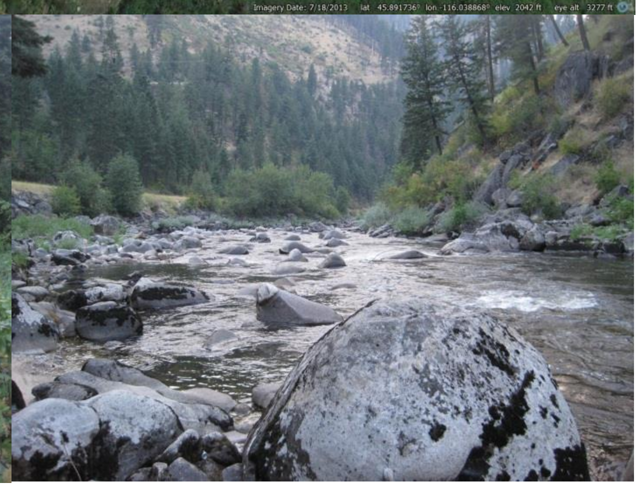
WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.912857	-116.005359	45m
Lower End:	45.913185	-116.005494	

Bottom of Site Looking Up

Top of Site Looking Down

Appendix H. SFCW 38.5 site is located approximately 0.25 road miles downstream of the Mt. Idaho Grade Road/Highway 14 junction. There is a pullout on the side of the highway adjacent to the river. The site is right next to the pullout and consists of pocket water. **Water can be swift and potentially dangerous due to substrate, take caution when snorkeling**. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.891298	-116.039152	30m
Lower End:	45.891392	-116.039491	

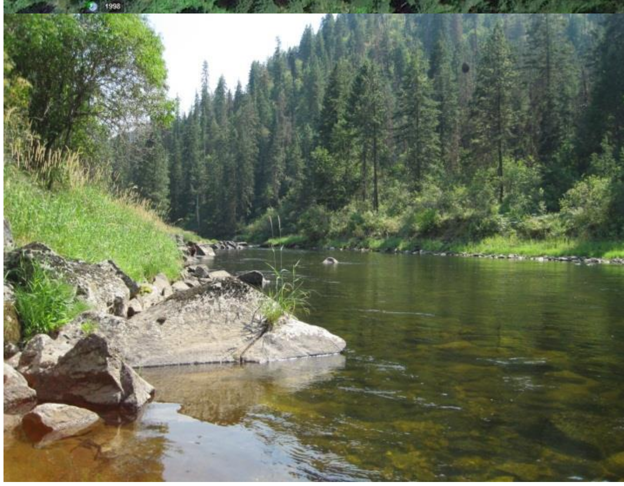


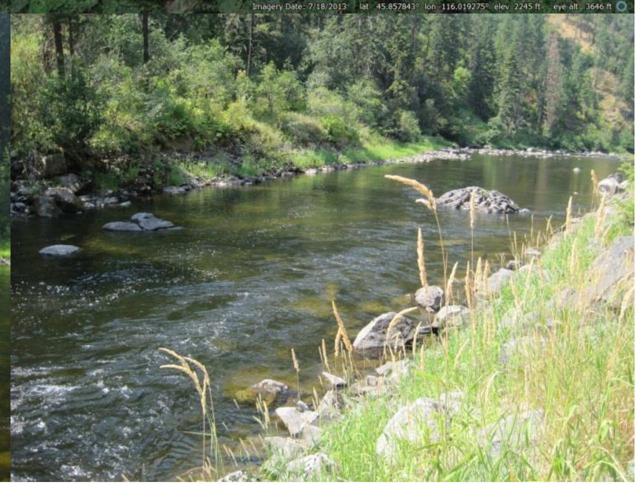
Bottom of Site Looking Up

Top of Site Looking Down

Appendix I. SFCW 43.9 The McAllister Picnic Area is about 2.8 road miles upstream of the Mt. Idaho Grade Road (NF-17)/Highway 14 junction and provides a good place to park. The site is approximately 300 meters downstream of the picnic area (just around the bend). The site is a run with a narrow riffle on the upstream side and a broad, shallow riffle downstream. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.859381	-116.019754	116m
Lower End:	45.860495	-116.020651	

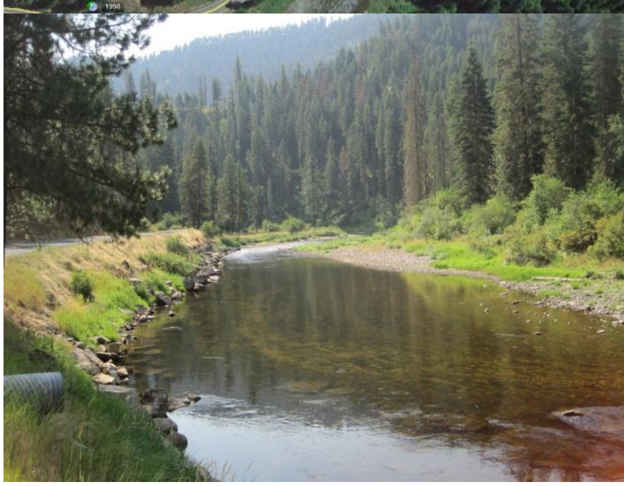




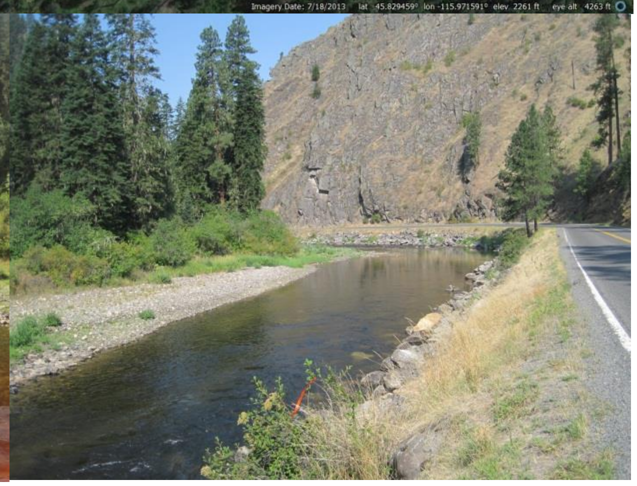
Bottom of Site Looking Up Top of Site Looking Down

Appendix J. SFCW 48.7 The Castle Creek Campground is approximately 6.1 miles upstream of the Mt. Idaho Grade Road/Highway 14 junction and provides a good place to park. The site is just downstream of the campground. The site is a run with a riffle above and below. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.829053	-115.970268	142m
Lower End:	45.830112	-115.971371	



Bottom of Site Looking Up



Top of Site Looking Down

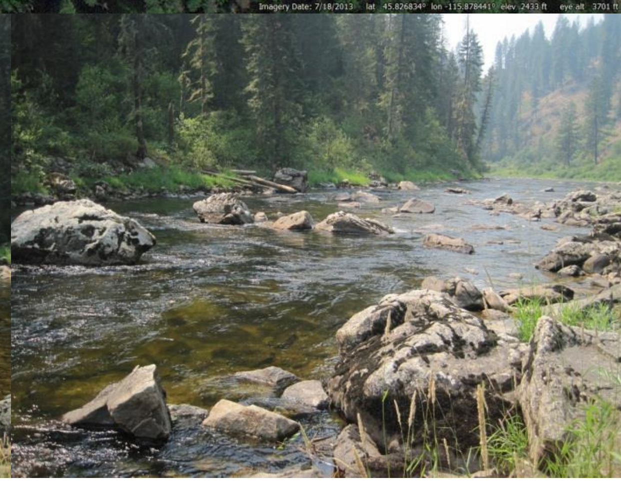
Appendix K. SFCW 53.0 there is a pullout just upstream of the mouth of Meadow Creek approximately 8.4 miles upstream of the Mt. Idaho Grade Road/Highway 14 junction for parking. The site is a run right next to the pullout. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.827462	-115.927002	163m
Lower End:	45.82819	-115.928227	


Bottom of Site Looking Up Top of Site Looking Down

Appendix L. SFCW 58.2 There is a large pullout approximately 11.1 miles upstream of the Mt. Idaho Grade Road/Highway 14 junction for parking. The site is approximately 150 meters upstream from the pullout and consists of mostly pocket water. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.826564	-115.876777	52m
Lower End:	45.826671	-115.877458	

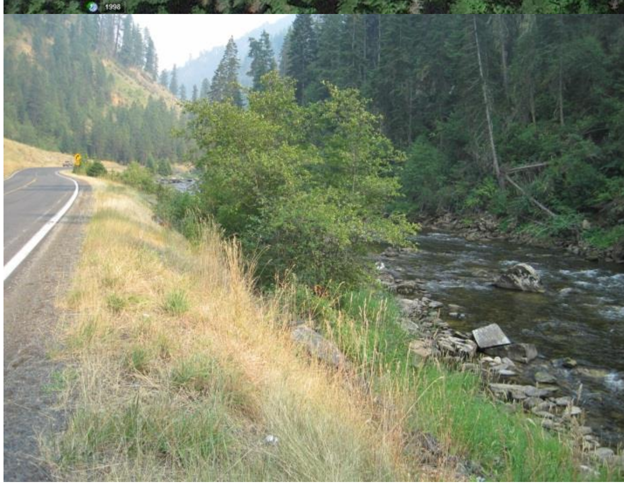


Bottom of Site Looking Up

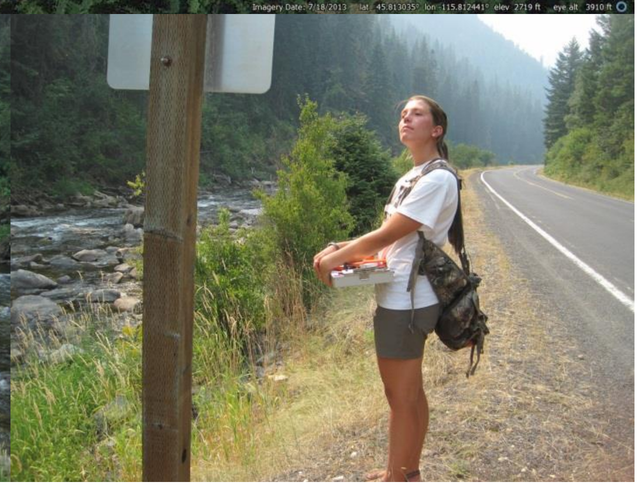
Top of Site Looking Down

Appendix M. SFCW 63.7 site is approximately 14.7 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction. There is a small pullout on the side of the road adjacent to the river approximately 0.4 miles upstream of Peasley Creek Road. The site is next to the pullout and consists mainly of pocket water. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.812471	-115.811467	106m
Lower End:	45.812849	-115.81267	



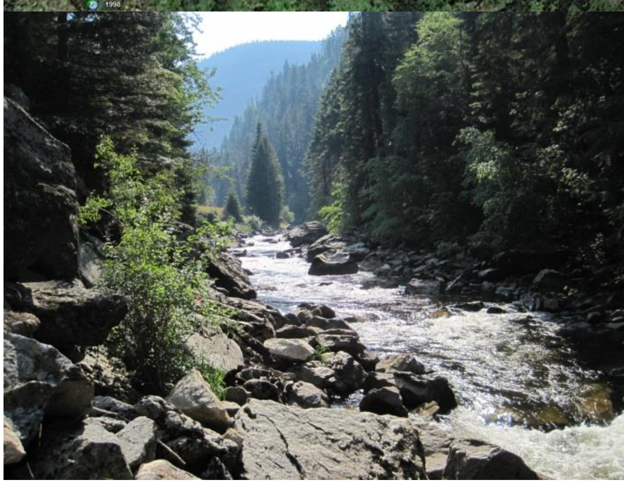
Bottom of Site Looking Up



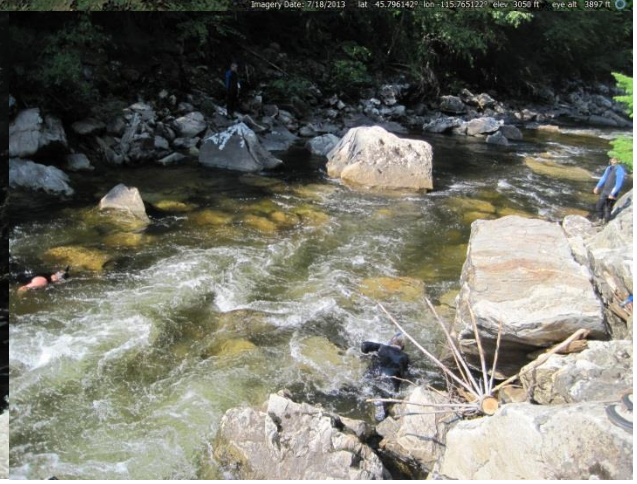
Top of Site Looking Down

Appendix N. SFCW 68.6 site is approximately 17.9 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction. There is a pullout on the side of the highway adjacent to the river. The site is next to the pullout and consists of a run with pocket water. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.795391	-115.764055	39m
Lower End:	45.79545	-115.764513	



Bottom of Site Looking Up



Top of Site Looking Down

Appendix O. SFCW 73.7 approximately 20.7 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction there is a large pullout for parking on the side of the highway adjacent to the river. The site is approximately 200 meters upstream of the pullout and consists of a run with pocket water. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.800921	-115.708716	43m
Lower End:	45.800632	-115.709278	

Bottom of Site Looking Up Top of Site Looking Down

Appendix P. SFCW 78.3 site is approximately 23.6 road miles upstream of the Mt. Idaho Grade Road/ Highway 14 junction. There is a pullout on the side of the highway adjacent to the river approximately 0.65 miles downstream of Buckhorn Road. The site is just upstream of the pullout and consists of a run with pocket water. Site was snorkeled downstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.809951	-115.663257	68m
Lower End:	45.810105	-115.663989	

Bottom of Site Looking Up

Top of Site Looking Down

Appendix Q. SFCW 83.9 the site is approximately 26.7 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction. There is a pullout on the side of the highway adjacent to the river just upstream of the bridge for road NF-492. The site is a shallow run with a riffle on either end. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.823746	-115.633792	99m
Lower End:	45.823118	-115.634641	

Bottom of Site Looking Up

Top of Site Looking Down

Appendix R. SFCW 88.7 the site is approximately 29.1 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction. There is a large pullout on the side of the highway adjacent to the river approximately 0.7 miles upstream of Newsome Bridge Road. The site is a run with a riffle on either end. Site was snorkeled downstream in 2014.

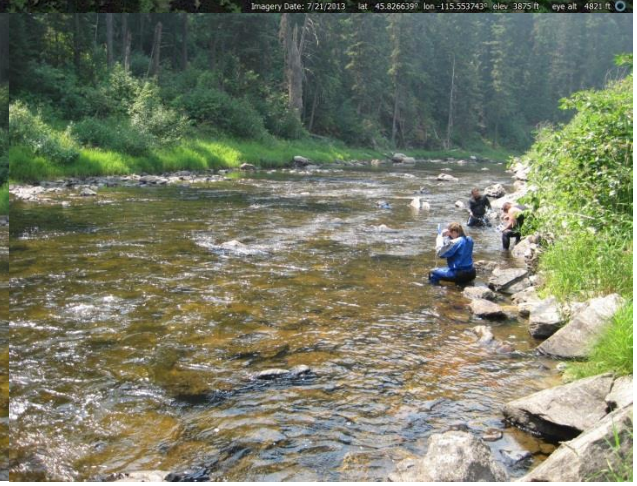
WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.823441	-115.596066	32m
Lower End:	45.823813	-115.596044	

Bottom of Site Looking Up

Top of Site Looking Down

Appendix S. SFCW 93.9 Pullout for parking is approximately 32.7 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction on the side of the highway adjacent to the river. The pullout is approximately 1.7 miles downstream of Crooked River Road. The site is 180 meters upstream of the pullout, just around the bend. The site consists of a shallow run. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.825641	-115.552539	44m
Lower End:	45.825752	-115.553032	

Bottom of Site Looking Up Top of Site Looking Down

Appendix T. SFCW 98.7 Site is approximately 35.2 road miles upstream of the Mt. Idaho Grade Road/Highway14 junction. There is a pullout on the side of the highway opposite of the river upstream of a couple houses and an old Inn. The site is 160 meters upstream of the pullout and consists of a shallow run with some pocket water. Site was snorkeled downstream in 2014.

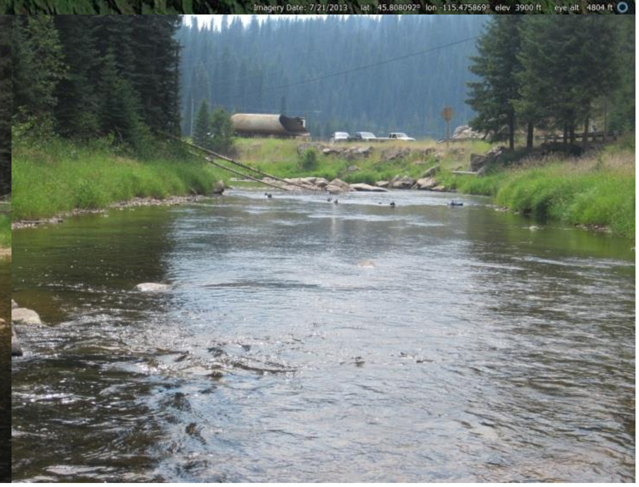
WGS84	Latitude	Longitude	Length of Site
Upper End:	45.825761	-115.508692	67m
Lower End:	45.826036	-115.509334	

Bottom of Site Looking Up

Top of Site Looking Down

Appendix U. SFCW 103.2 Site is approximately 38 road miles upstream of the Mt. Idaho Grade Road/Highway 14 junction. There is a large pullout on the side of the highway adjacent to the river 140 meters downstream of Red River Road. The upper end of the site is the confluence of the Red and American Rivers. Site was snorkeled upstream in 2014.

WGS84	**Latitude**	**Longitude**	**Length of Site**
Upper End:	45.808075	-115.475465	120m
Lower End:	45.807349	-115.476506	



Bottom of Site Looking Up

Top of Site Looking Down

MOUNTAIN LAKES MONITORING IN CONSIDERATION OF AMPHIBIAN RISK ASSESSMENT IN NORTH-CENTRAL IDAHO

ABSTRACT

We conducted the eleventh year of a 20-year study evaluating whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations. Preliminary analysis suggests gill net CPUE has declined over time in Middle Wind, Hungry, and Siah lakes; however, there was a stable trend in CPUE across all project lakes containing fish. For high mountain lakes surveyed in 2017, there was a stable trend in long-term Columbia spotted frog *Rana luteiventris* (CSF) and long-toed salamander *Ambystoma macrodactylum* (LTS) presence. Long-toed salamander presence was negative correlated with the habitat variable “fish presence”, while CSF presence was not correlated with any habitat variables. Both species were positively correlated with the temporal variable “Julian Date^2”. For surveys conducted from 2014 to 2017, the composite detection probability for all life stages detected during visual encounter surveys was 0.92 for CSF and 0.62 for LTS. Declining gill net CPUE could be a result annual variation in weather at the time of the survey, or broader drivers such as reduced food resources or recruitment failure caused by severe winter or summer conditions. Long-term trends in CSF and LTS presence were consistent with previous findings and indicate that these populations have remained stable throughout the duration of this study. The negative relationship between LTS and fish presence aligns with previous findings in this study, and other studies conducted throughout their range. Based on the preliminary trends in amphibian populations with our study lakes, current fisheries management of high mountain lakes appears adequate for balancing fishing opportunity with the long-term persistence of amphibians. However, a more detailed analysis at the end of this study will be necessary to determine impacts on a larger scale, and especially in HUCs that include lakes currently within the IDFG high mountain lake stocking program.

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INTRODUCTION

Amphibian population reduction and species extinction has given urgency to amphibian conservation, inventory efforts to determine baseline data, and monitoring to determine trends in amphibian populations (Houlahan et al. 2000; Stuart et al. 2004; Beebee and Griffiths 2005; Orizaola and Brana 2006). Potential factors in amphibian population decline are numerous and include: habitat modification/fragmentation, introduction of predators/competitors, increased UV-B radiation, changes in precipitation/snowpack, and pathogen infection (Alford and Richards 1999; Corn 2000; Marsh and Trenham 2001; Pilliod and Peterson 2001). Throughout the North Central Mountains of Idaho, direct (predation) and indirect (resource competition, habitat exclusion, and population fragmentation) impacts on amphibian populations from introductions of trout into historically fishless lakes are also a cause for concern (Semlitsch 1988; Figiel and Semlitsch 1990; Bradford et al. 1993; Brönmark and Edenhamn 1994). Trout have been stocked into high mountain lakes to provide recreational opportunities to backcountry visitors. As much as 95% of previously and/or currently stocked high mountain lakes throughout the western United States that were once fishless, now contain fish through regular stocking efforts or self-sustaining populations from legacy stocking efforts (Bahls 1992). It is estimated that 96% of lakes within the Nez Perce-Clearwater National Forest were historically fishless as the headwater area topography where lakes are located is relatively steep (Murphy 2002). According to historical stocking records, some lakes in North Central Idaho were stocked as early as the 1930s (Murphy 2002). Out of the estimated 3,000 mountain lakes in Idaho, approximately 1,355 lakes (45%) are stocked or have natural fish populations (IDFG 2012).

Mountain lake ecosystems in North Central Idaho contain amphibians such as long-toed salamander *Ambystoma macrodactylum* (LTS) and Columbia spotted frog *Rana luteiventris* (CSF), although Idaho giant salamander *Dicamptodon aterrimus*, western toad *Bufo boreas*, and Rocky Mountain tailed frog *Ascaphus montanus* may also be present. Common reptiles found at these mountain lakes may also include common Garter snake *Thamnophis sirtalis* and western terrestrial Garter snake *T. elegans*, both of which were historically (before fish introductions) the main amphibian predators (Murphy 2002).

Surveys have found that CSF occurrence (and breeding occurrence) in the Clearwater Region was not significantly different in lakes with or without fish after accounting for habitat effects (CSF were positively associated with increasing amounts of sedge meadow perimeter and silt/organic substrate) (Murphy 2002). However, CSF abundance at all life stages was significantly lower in lakes with fish than without fish (Murphy 2002). In contrast, LTS larvae and/or breeding adult occurrence and abundance (adults are typically terrestrial except to breed) was significantly less common in lakes with fish then lakes without fish (Murphy 2002). However, where native (not stocked) Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) existed in lakes, the impact on LTS was not as severe as compared to lakes that were historically fishless and later stocked with trout (Murphy 2002). Other studies have examined relationships between introduced trout and salamanders. Direct negative impacts by fish on amphibian populations have been mostly attributed to trout preying upon amphibians when they are at a larval stage, although trout may also cause salamanders to avoid lakes previously used as breeding sites (Kats et al. 1993; Figiel and Semlitsch 1990; Bradford et al. 1993; Knapp 1996; Pilliod et al. 1996; Graham and Powell 1999; Murphy 2002).

Introduced fish populations may also indirectly impact amphibian gene flow, recolonization, and subsequent persistence. The degree of gene flow in mountain lake amphibians likely relies on connectivity between higher and lower elevations subpopulations (with low gene flow). Gene flow may also occur between neighboring lakes that are not necessarily within the same wet stream migration corridor when overland dispersal is not drastically limited by headwater topography, precipitation, and or canopy cover (Murphy 2002). Long-toed salamander within north-central Idaho are panmictic (randomly interbreeding populations), with high levels of within population variation providing evidence that populations are not evolving in complete isolation (Tallmon et al. 2000). Amphibian populations or demes in these headwater areas likely never evolved with native fish and may lack the appropriate defensive, behavioral, or chemical responses to coexist with introduced fish populations (Kats et al. 1988).

Westslope Cutthroat Trout, Rainbow Trout *O. mykiss* (RBT), RBT x WCT hybrids, and Brook Trout *Salvelinus fontinalis* (BKT) are the most common introduced fish species in high mountain lakes in the Clearwater Region. Although, many lakes within the study area have a stocking history that may include Yellowstone Cutthroat Trout *O. bouvieri*, California Golden Trout *O. m. aguabonita* (last stocked in 1990 in the Clearwater Region - Steep Lakes), Arctic Grayling *Thymallus arcticus* (last stocked in 1982 in the Clearwater Region - Bald Mountain Lake), and various forms of trout hybrids. The term “introduced western trout” may be more appropriate for *Oncorhynchus* species in these lakes where natural reproduction is occurring, as the degree of hybridization is unknown in lakes where multiple species have been stocked (Behnke 1992). The Clearwater Region currently stocks 87 of its 711 high mountain lakes. Most lakes are stocked with fingerling WCT on a three-year rotation by fixed-wing aircraft.

Certain species of introduced trout tend to have a greater impact on amphibian occupancy than others (Murphy 2002). Brook Trout tend to impact CSF and especially LTS occurrence and breeding to a greater extent than the presence of WCT or RBT. This impact is derived from differences in fish spawning times/behavior and variations in amphibian habitat usage just after ice off conditions in mountain lakes (Murphy 2002). Westslope Cutthroat Trout and RBT in these lakes spawn in spring/summer which often coincides with times that amphibian breeding occurs. As a result, both fish species are typically preoccupied with spawning in inlets or outlets while amphibians are typically breeding within the lake itself. This difference in spawning habitat use may allow amphibians to breed with fewer disturbances by WCT and RBT (Murphy 2002). In contrast, BKT are fall spawners and are actively moving and foraging throughout the lake in spring and are more likely to prey upon any amphibian life stage and/or harass breeding adults (Murphy 2002). Furthermore, BKT tend to be more benthic oriented (where salamanders usually occur), utilize larger prey items, and attain higher densities within mountain lakes than *Oncorhynchus* species (Griffith 1974). Columbia spotted frog do not tend to be impacted by BKT presence to the same magnitude as LTS because of their different habitat associations and shorter length of larval stages (Griffith 1974; Bahls 1992; Murphy 2002).

Long-toed salamander occupies a wide range over the western United States and Canada. The majority of LTS in Idaho sub-alpine lakes have a two year larval stage, making them susceptible to predation by fish for a longer period of time. Studies suggest that they are more susceptible to impacts by introduced fish than the CSF (Murphy 2002). Conclusive evidence of LTS decline is insufficient (Graham and Powell 1999). For this reason, a long-term monitoring project (20 years) was initiated in the Clearwater Region to provide knowledge of the amphibian population dynamics within the north-central mountains of Idaho. Long-term monitoring of mountain lakes will allow for amphibian population trends to be identified and will give managers the ability to determine whether sufficient fishless habitat exists to support amphibian populations into the future.

OBJECTIVES

1. Evaluate whether high mountain lakes within the IDFG Clearwater Region that have fish are less likely to have amphibians.

2. Assess whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations.

STUDY AREA

The 74 lakes selected for this study are located within the Nez Perce-Clearwater National Forests, located in north-central Idaho (Figure 70). High mountain lakes within this study are primarily located in wilderness areas (Selway-Bitterroot, Gospel Hump, and Frank Church River of No Return Wilderness) within the Nez Perce-Clearwater National Forest, with two lakes located outside of wilderness boundaries. There are nine HUC5 creek drainages in the study: Goat, Upper Meadow, Big Harrington, North Fork Moose, Storm, Running, Warm Springs, Old Man, and Bargamin creeks. Hiking and multi-day backpacking trips are required to access all of the high mountain lakes in this study.

METHODS

The study design and protocol for this long-term evaluation utilized an amphibian risk assessment model developed through previous studies and inventories of mountain lakes conducted within north-central Idaho. This model is based on the amount of fishless habitat that exists within a watershed at the HUC5 level. At the individual HUC5 watershed level, it is assumed monitoring will be able to examine conditions that may dictate local response in the interactions of stocked fish and native amphibian populations to provide a more defined opportunity for prioritized management action (Murphy 2002). While there are many risk factors associated with amphibian declines, our assessment focused on considering impacts that may be associated with native and stocked fish in lakes on a HUC5 watershed basis. The amphibian risk assessment model for these high mountain lake ecosystems has four categories: control (no risk), low, moderate, and elevated.

- *Control or no risk* – watershed has never experienced fish introductions through stocking activities.
- *Low* – At least 50% of the lakes within a watershed are fishless <u>AND</u> a minimum 20% of the lake surface area within the watershed is fishless.
- *Moderate* – 50% of lakes within a watershed are fishless <u>OR</u> 20% of surface area is fishless.
- *Elevated* – Meets neither requirement, less than 50% of the lakes within a watershed are fishless <u>AND</u> less than 20% of the surface area within the watershed is considered fishless.

Two HUC5 watersheds were selected randomly from each of the amphibian risk categories (region-wide from all HUC5 watersheds that contained lakes) for sampling. This resulted in eight HUC5 watersheds containing 72 lakes within the Nez Perce-Clearwater National Forest. In 2013, a third randomly selected HUC5 watershed (Big Harrington Creek) was added to increase the sample size of fishless control lakes, bringing the study's total to nine watersheds that contain 74 lakes (Hand et al. 2016). Attempts will be made to sample all lakes within a selected HUC5 watershed within the same field season. The 20-year period for the high mountain lakes long-term monitoring project will allow for each of these lakes be sampled ~6 times each. The repetition of sampling events will allow for comparisons to be made within (for trends) and between watersheds (for comparisons among amphibian risk classes). In addition, repetition of sampling events will address the normal patterns of recruitment fluctuations often common among amphibian populations. Sampling frequency and rotation order are adjusted as needed due to weather, trail, and fire conditions.

Field sampling

In 2017, 30 lakes were selected to be surveyed within the North Fork Moose Creek, Storm Creek, and Warm Springs Creek HUC5 units (Figure 70). Field sampling was conducted following the protocol used throughout the duration of this project, and revised after the 2013 field season to improve the accuracy and comparability of results from year-to-year (Hand et al. 2016). Beginning with the 2014 field season we introduced the use of multiple VES at a lake within a 24-hour timeframe to increase the probability of detection for amphibians, and to allow for estimating detection probabilities. These surveys were separated by at least three hours and conducted during different parts of the day (i.e. morning and afternoon), or even different days, when time allows.

Data analysis

We evaluated trends in gill net CPUE (fish/h) across all project lakes containing fish using least squares regression with survey number (e.g. 1st survey conducted, 2nd survey, etc.) as the independent variable and $\log_e$ transformed CPUE as dependent variables (Maxell 1999; Kennedy and Meyer 2015). We did not evaluate trends in CPUE based on year as lakes are not surveyed in the same year, or at consistent intervals. The rate of change (r_{intr}) in CPUE is determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance, where the trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$.

To assess the impacts of fish presence on amphibian populations, we evaluated long-term trends in amphibian presence, factors influencing amphibian presence, and detection probabilities by comparing data collected from 1988 to 2017. Trends in amphibian populations were evaluated for 56 project lakes with both historic data (collected prior to this study) and three rounds of surveys within this study. These lakes were selected for this report to provide the longest time series available. A full evaluation including all project lakes will be conducted at the end of the study. The methods used for statistical analysis are explained in detail in Hand et al. (2016), and Hand et al. (2018). We utilized R Studio (R version-3.5.1 (2018-07-02) “Feather Spray”) to conduct the statistical analyses for this report (R Core Team 2014).

RESULTS

Fish surveys

Fish were present in 20% of the lakes surveyed in 2017 (Table 30). Gill net CPUE (fish/h) ranged from 0.2 to 5.0/h (Table 31). Gill net CPUE has been stable (r_{intr} = -0.045; 90% CI bounds

= -0.211 - 0.120). However, CPUE has declined in Middle Wind, Hungry, and Siah lakes over the course of this study (Figure 71).

Amphibian Surveys

Columbia spotted frog

Columbia spotted frog was observed in 93% of lakes surveyed (Table 30). Through 2017, 56 of the 74 project lakes now have historic data and three rounds of data from this project. For these lakes, there was no trend in CSF presence (Figure 72). Of the 28 lakes where CSF were observed, 21% contained fish.

Occurrence of all life stages of CSF (presence/absence) was positively correlated with "Julian Date", but negatively correlated with "Julian Date^2" (Table 32). "Fish Presence" was not correlated with CSF presence (Table 32). Over the course of this study, observations of CSF were consistent through the sampling season, but decline in early September (Figure 73).

The detection probability for all life stages of CSF was 0.90 for multiple VES surveys. For surveys conducted from 2014 to 2017, the composite detection probability for CSF was 0.92.

Long-toed salamander

Long-toed salamander was observed in 70% of lakes surveyed (Table 30). Through 2017, 56 of the 74 project lakes now have both historic data and three rounds of data from this project. For these lakes, there was no trend in CSF presence (Figure 72). Of the 21 lakes where LTS were observed, 10% contained fish.

Occurrence of all life stages of LTS (presence/absence) was positively correlated with "Julian Date", but negatively correlated "Julian Date^2" and "Fish Presence" (Table 32). Over the course of this study, no observations of LTS have occurred beyond mid-September (Figure 73).

The detection probability for all life stages of LTS was 0.62 for multiple VES surveys. For surveys conducted from 2014 to 2017, the composite detection probability for LTS was 0.62.

DISCUSSION

Fish surveys

Preliminary analysis suggests gill net CPUE has declined over time in Middle Wind, Hungry, and Siah lakes; however, there was no statistically significant trend in CPUE across all project lakes containing fish. Declining CPUE could be a result annual variation in weather at the time of the survey, or broader drivers such as reduced food resources or recruitment failure caused by severe winter or summer conditions (Armstrong and Knapp 2004; Parker et al. 2008). A concern was that our periodic surveying has suppressed fish populations in lakes with low productivity, poor recruitment, and/or slow growth (Gray 2013). With declining CPUE occurring in only a few lakes, suppression from our sampling is not likely the cause. However, this is worth monitoring as we conduct additional surveys. A more detailed analysis of factors that may be influencing CPUE is beyond the scope of this report, but should be evaluated further if we continue to observe declining CPUEs in some lakes.

Amphibian surveys

Columbia spotted frog

Our preliminary analysis indicates CSF presence has remained stable throughout the duration of this study. This was consistent with previous findings (Hand et al. 2020). At this time, the data suggest that any impacts on amphibian populations by fish have likely already occurred, and they have reached a point of stability and are not in a continued state of decline (Sexton and Phillips 1986; Knapp and Matthews 2000; Pilliod and Peterson 2001). Further analysis will be conducted after additional surveys have been completed.

Columbia spotted frog presence was not correlated with any habitat variables, including “Fish Presence”. Habitat variables such as emergent vegetation and percent littoral zone have often been correlated with CSF presence and abundance, especially in lakes containing fish (Pilliod and Peterson 2001; Bull and Marx 2002). The lack of correlation in our study is likely due to the presence of CSF at 93% of study lakes, including 93% of lakes that contain fish. At every lake where fish were present there was concurrent presence of CSF. This is important to consider within the framework of this study, as it appears historic fish stocking is not having a continued impact on CSF populations.

In contrast, the temporal variable “Julian Date^2” was a significant predictor of CSF presence. Variation in detection related to Julian date has been explained by within-year variation in microclimate and localized growth conditions that cause amphibians to metamorphose at different times in different sub-basins (Pilliod and Peterson 2001). Another explanation for daily variation in detection rates is that weather directly influences the efficacy of the VES. Amphibians may be less active, and therefore less noticeable, during colder weather conditions or storms (Lohr and Haak 2011). Therefore, yearly variation in weather, snow conditions, and even day to day conditions within a given high mountain lake will influence the observable amphibian populations. Given considerations of amphibian population age structure, weather, and individual surveyor bias, VES surveys should continue to be performed twice at each lake to improve likelihood of detection.

The composite detection probability for all life stages of CSF was 0.92 for multiple VES surveys conducted from 2014 to 2017. Annual detection probabilities have stayed within a small range (0.89 to 0.95). This indicates that our sampling has remained consistent on an annual basis, and that detection probabilities are similar annually in spite of the potential for biases associated with differences in weather, lakes visited each year, and observers.

Long-toed salamander

Our preliminary analysis indicates LTS presence has remained stable throughout the duration of this study. This was consistent with previous findings (Hand et al. 2020). At this time, the data suggest that any impacts on amphibian populations by fish have likely already occurred, and they have reached a point of stability and are not in a continued state of decline (Sexton and Phillips 1986; Knapp and Matthews 2000; Pilliod and Peterson 2001). Further analysis will be conducted after additional surveys have been completed.

Long-toed salamander presence was negatively correlated to fish presence. This relationship aligns with previous findings in this study, and other studies conducted throughout their range (Murphy 2002; Pearson and Goater 2009; Hand et al. 2016; Kenison et al. 2016). Since the beginning of this study, LTS have been observed at least once in 95% of all fishless lakes within this study, but only 48% of lakes containing fish. The negative impact of fish on LTS presence has been attributed to their two-year larval stage, where they have longer exposure to fish predation during their vulnerable aquatic rearing (Pilliod and Peterson 2001; Pearson and Goater 2009). Greater overall predation on larval LTS relative to larval CSF likely explains why CSF can co-occur with fish in lakes where LTS are not detected or are detected only in low densities. Although LTS abundance and distribution may have stabilized, fish presence has a direct impact on this species.

As with CSF, the temporal variable "Julian Date^2" was a significant predictor of LTS presence. Variation in detection related to Julian date has been explained by within-year variation in microclimate and localized growth conditions that cause amphibians to metamorphose at different times in different sub-basins (Pilliod and Peterson 2001). Another explanation for daily variation in detection rates is that weather directly influences the efficacy of the VES. Amphibians may be less active, and therefore less noticeable, during colder weather conditions or storms (Lohr and Haak 2011). Therefore, yearly variation in weather, snow conditions, and even day-to-day conditions within a given high mountain lake will influence the observable amphibian populations. Given considerations of amphibian population age structure, weather, and individual surveyor bias, VES surveys should continue to be performed twice at each lake to improve likelihood of detection.

The composite detection probability for all life stages of LTS was 0.62 for multiple VES surveys conducted from 2014 to 2017. Annual detection probabilities have stayed within small ranges (0.55 to 0.65). This indicates that our sampling has remained consistent on an annual basis, and that detection probabilities are similar annually in spite of the potential for biases associated with differences in weather, lakes visited each year, and observers.

Conclusion

Based on the preliminary trends in amphibian populations with our study lakes, current fisheries management of high mountain lakes appears adequate for balancing fishing opportunity with the long-term persistence of amphibians. However, a more detailed analysis at the end of this study will be necessary to fully evaluate this stocking program to ensure it meets the needs of both anglers and amphibian conservation.

MANAGEMENT RECOMMENDATIONS

1. Continue evaluating whether high mountain lakes within the IDFG Clearwater Region that have fish are less likely to have amphibians.

2. Continue to assess whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations.

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Table 30. Presence and counts of Columbia spotted frog (CSF) and long-toed salamander (LTS), and fish presence, in high mountain lakes surveyed in the Clearwater Region, Idaho, during 2017.

Lake	CSF present	LTS present	Fish Present	CSF adults	CSF larvae	LTS adults	LTS larvae	Risk Assessment
North Wind (upper)	no	no	no	0	0	0	0	Moderate
North Wind (lower)	no	no	no	0	0	0	0	Moderate
Middle Wind	yes	yes	yes	38	0	1	0	Moderate
South Wind	yes	yes	no	11	1	1	36	Moderate
North West Wind	yes	yes	no	5	1	0	2	Moderate
West Wind	yes	yes	yes	2	1	0	1	Moderate
East Wind	yes	no	yes	20	3	0	0	Moderate
Wind Pond	yes	yes	no	4	1	0	67	Moderate
Dodge WSC	yes	yes	no	35	35	2	0	Moderate
Hungry	yes	no	yes	29	1	0	0	Moderate
North Section 25	yes	yes	no	13	15	2	0	Low
South Section 25	yes	yes	no	30	2	0	1	Low
Old Stormy	yes	yes	no	12	7	2	73	Low
Middle Storm	yes	yes	no	41	10	0	50	Low
North Storm	yes	yes	no	1	0	0	6	Low
Storm	yes	yes	no	2	0	0	66	Low
West Moose 1	yes	yes	no	821	695	7	348	Low
West Moose 2	yes	yes	no	639	1,418	0	10	Low
West Moose 3	yes	yes	no	51	162	0	2	Low
West Moose 4	yes	yes	no	301	2,060	0	9	Low
West Moose 5	yes	yes	no	32	649	0	6	Low
West Moose 6	yes	yes	no	71	43	0	24	Low
West Moose 7	yes	no	no	72	2	0	0	Low
West Moose 8	yes	no	no	58	21	0	0	Low
West Moose 9	yes	no	no	51	14	0	0	Low
Section 28	yes	yes	no	121	1,169	50	82	Low
Siah	yes	no	yes	185	246	0	0	Low
Section 27 SC	yes	yes	no	39	776	8	99	Low
N.E. Ranger	yes	yes	no	8	582	3	606	Low
Ranger	yes	no	yes	7	129	0	0	Low

Table 31. Catch-per-unit-effort (fish/h) for all gill net surveys conducted in Clearwater Region, Idaho, high mountain lakes surveyed in 2017.

	Survey number			
Lake	1	2	3	4
Hungry	3.66	0.70	1.75	
Ranger	0.54	0.35	0.33	
Siah	1.79	0.19	0.29	
East Wind	1.45	0.93	2.12	2.00
Middle Wind	3.95	4.22	2.39	2.90
West Wind	0.42	0.23		

Table 32. Variables and associated p-values for generalized linear model analysis of Columbia spotted frog and long-toed salamander presence in high mountain lakes surveyed in the Clearwater Region, Idaho, in 2017. Significance was set at α = 0.10.

Columbia spotted frog		Long-toed salamander	
Variable	P-value	Variable	P-value
Elevation	0.623	Elevation	0.464
Fish presence	0.919	Max depth	0.728
Max depth	0.704	Fines	0.927
Fines	0.172	Fish presence	< 0.001
Julian Date	< 0.001	Julian Date	< 0.001
Julian Date 2	< 0.001	Julian Date 2	< 0.001

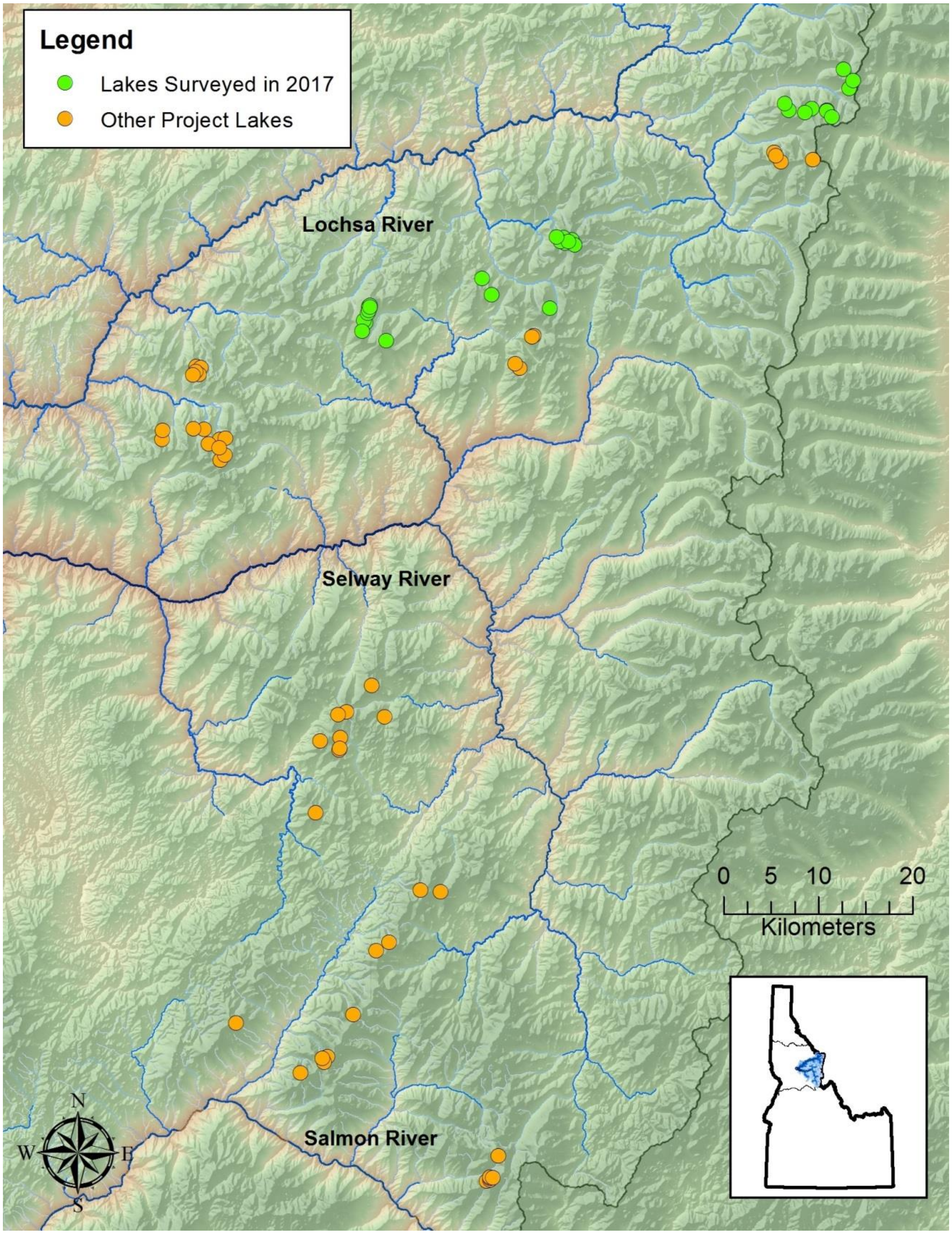

Figure 70. Map depicting high mountain lakes selected to evaluate long-term trends in amphibian populations in the Clearwater Region of Idaho, including those lakes surveyed during 2017.

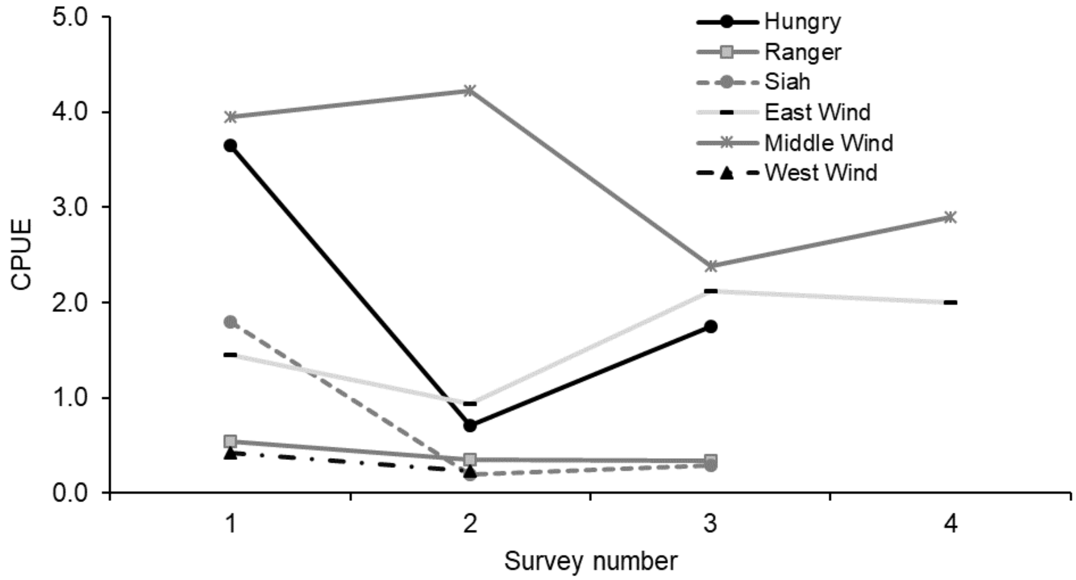

Figure 71. Catch-per-unit-effort for all gill net surveys conducted from 2006 to 2017 on high mountain lakes surveyed in the Clearwater Region, Idaho, in 2017.

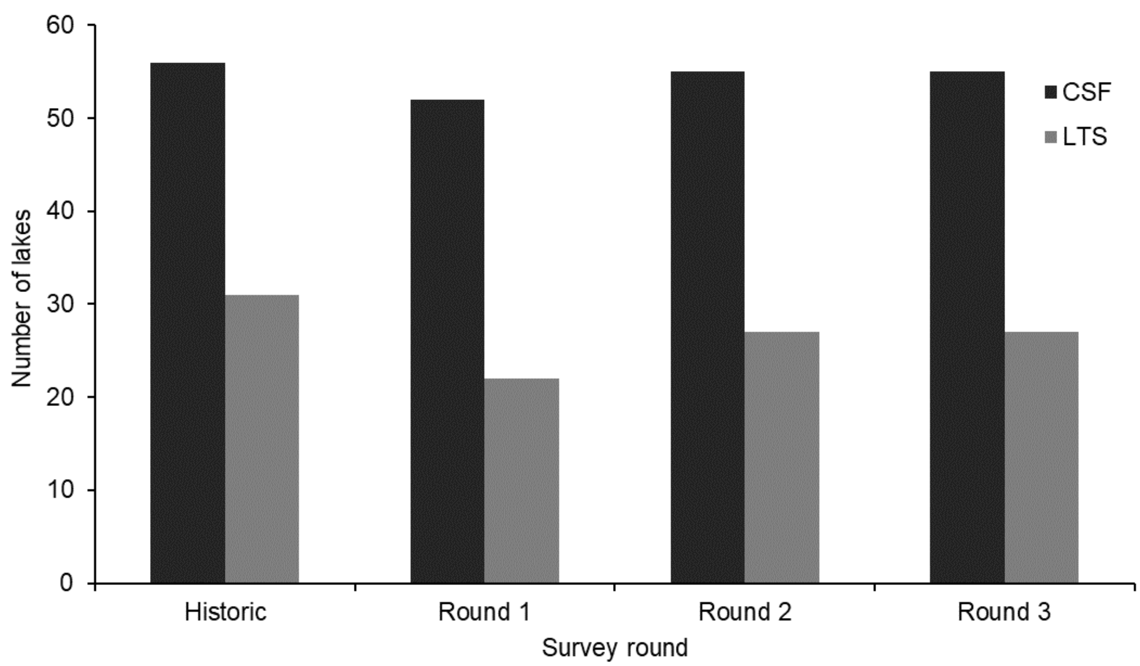

Figure 72. Presence of Columbia spotted frog (CSF) and long-toed salamander (LTS) in mountain lakes (n = 56) of the Clearwater Region, Idaho, which have historic data (prior to this project) and three rounds of project surveys completed.

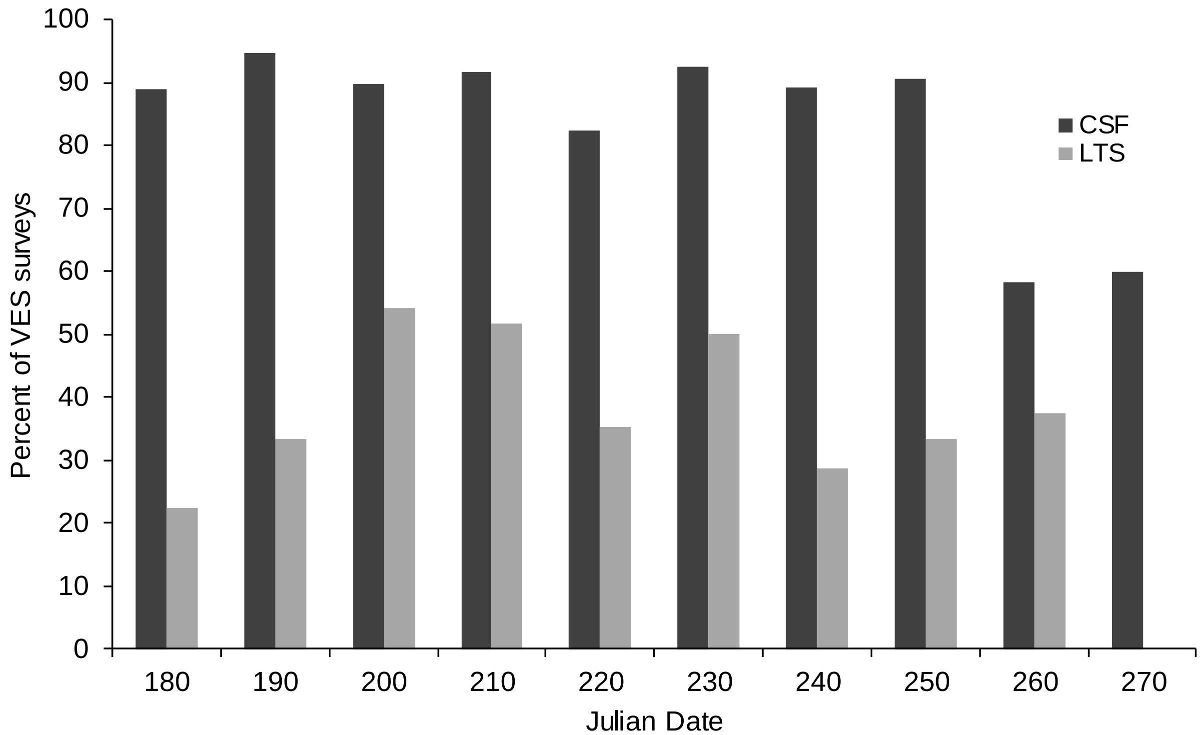

Figure 73. Percent of Visual Encounter Surveys (VES) with Columbia spotted frog and long-toed salamander present in high mountain lake surveys of the Clearwater Region, Idaho, conducted from 2006 to 2017, based on Julian Date.

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